

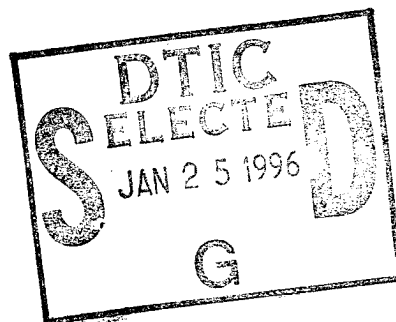


**US Army Corps
of Engineers**
Waterways Experiment
Station

Wetlands Research Program Technical Report WRP-SM-9

Interior Wetlands of the United States: A Review of Wetland Status, General Ecology, Biodiversity, and Management

by John H. Giudice, John T. Ratti



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| DE | Delineation & Evaluation | SM | Stewardship & Management |

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by John H. Giudice, John T. Ratti

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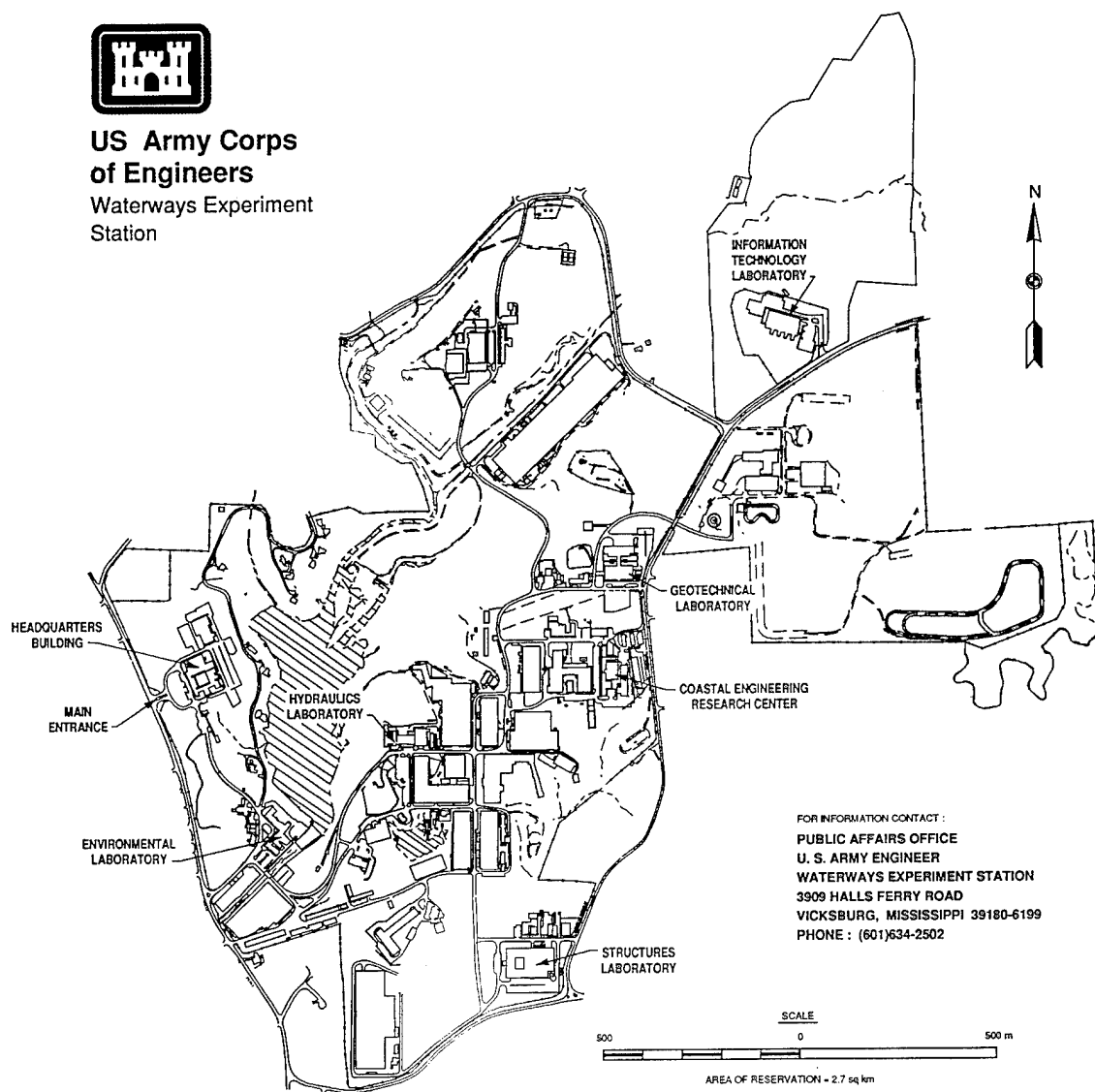
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Biodiversity and Natural Communities

Interior Wetlands of the United States: A Review of Wetland Status, General Ecology, Biodiversity, and Management (TR WRP-SM-9)

ISSUE:

Ecosystem management and the conservation of biological diversity (commonly termed biodiversity) has become an important public-policy issue in the United States. Although national attention has centered on terrestrial ecosystems, wetlands are an important component in efforts to conserve biodiversity. Freshwater wetlands support a wide diversity of plant and animal species, including a third of the nation's threatened and endangered species. Wetland managers are being asked to place more emphasis on biodiversity and natural community characteristics while simultaneously maintaining other wetland functions and values. Effectively meeting this challenge will require individuals who understand the concepts and principles of conservation biology and are capable of integrating this knowledge with more traditional information on ecology, management, status, and biopolitics of wetlands. Review and synthesis of available information on these subject areas will be valuable to wetland managers and policymakers.

RESEARCH:

This report was compiled to (a) provide Corps field and District-level personnel with a primer on the ecology, biodiversity, and management of freshwater wetlands in the United States; and (b) direct

interested personnel to more detailed sources of information on selected topics.

SUMMARY:

This report provides an overview of the principles, concepts, strategies, and techniques necessary to preserve, restore, create, and manage natural community and biodiversity characteristics on nontidal, freshwater wetlands of the United States. A brief review of wetland definitions, classification and inventory, status and distribution, general ecology, functions and values, and programs affecting wetland conservation is also provided.

AVAILABILITY OF REPORT:

The report is available on Interlibrary Loan Service from the U.S. Army Engineer Waterways Experiment Station (WES) Library, 3909 Halls Ferry Road, Vicksburg, MS 39180-6199; telephone (601) 634-2355.

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Contents

| | |
|--|----|
| Preface | x |
| 1—Introduction | 1 |
| 2—Defining Wetlands | 4 |
| Circular 39 Definition | 4 |
| Current USFWS Definition | 5 |
| Regulatory and Statutory Definitions | 5 |
| Riparian Ecosystems | 6 |
| 3—Wetland Classification | 8 |
| Conceptual and Semantic Problems | 8 |
| Classification Schemes | 9 |
| Martin et al. classification | 10 |
| Cowardin et al. classification | 10 |
| Brinson's hydrogeomorphic classification | 15 |
| Mitsch and Gosselink | 18 |
| 4—Inventory and Status of Wetlands | 19 |
| Inventory and Monitoring | 19 |
| Remote sensing | 19 |
| National wetland inventories | 20 |
| Status of Wetlands | 21 |
| Distribution | 21 |
| Abundance | 21 |
| Status and trends | 22 |
| Influences and Alterations | 24 |
| Human influences | 24 |
| Regional influences | 24 |
| Wetland alterations | 29 |
| 5—Wetland Ecology | 32 |
| Geomorphology and Hydrology | 32 |
| Wetland Soils | 35 |
| Biogeochemistry | 36 |

| | |
|---|----|
| Biological Adaptations | 38 |
| Biotic Components | 40 |
| Nutrient Cycling | 41 |
| Wetland Dynamics | 43 |
| Wetland Functions and Their Value | 45 |
| Wetland functions | 45 |
| Value of wetland functions | 47 |
| Valuation methods | 47 |
| 6—Biodiversity | 50 |
| Biodiversity: A Primer | 51 |
| Genetic diversity | 51 |
| Species diversity | 55 |
| Ecosystem diversity | 56 |
| Landscape diversity | 57 |
| Ecological and evolutionary processes | 58 |
| Value of Biodiversity | 58 |
| Status of Biodiversity | 60 |
| Wetland and Deepwater Habitats | 61 |
| Conservation Strategies | 66 |
| 7—Stewardship and Management | 68 |
| The Concept of Management | 68 |
| Managed Wetland Types | 72 |
| Freshwater marshes | 73 |
| Moist-soil impoundments | 74 |
| Greentree reservoirs | 75 |
| Management Techniques | 77 |
| Water-level manipulation | 77 |
| Prescribed burning | 81 |
| Control of herbivores | 81 |
| Control of exotic and other nuisance species | 82 |
| Mechanical treatment of vegetation and substrate | 82 |
| Chemical treatment of vegetation | 83 |
| Artificial nesting structures | 84 |
| Limits of Management | 84 |
| 8—Wetland Conservation and Protection | 87 |
| Regulatory Programs | 88 |
| Acquisition and Incentive Programs | 89 |
| Migratory bird hunting and conservation stamps (1934) | 89 |
| The Land and Water Conservation Fund Act (1965) | 89 |
| Water Bank Program | 92 |
| Other Federal programs | 93 |
| State and private programs | 93 |
| Restoration Programs | 94 |

| | |
|--|-----|
| 9—Summary | 95 |
| References | 100 |
| Appendix A: Common and Scientific Names of Plants | |
| Named in Text | A1 |
| Appendix B: Common and Scientific Names of Animals | |
| Named in Text | B1 |
| Appendix C: Selected Readings | C1 |
| SF 298 | |

List of Figures

| | |
|--|----|
| Figure 1. Wetlands are often located at the ecotones between terrestrial systems and permanently flooded deepwater-aquatic systems such as rivers, lakes, estuaries, or oceans | 2 |
| Figure 2. U.S. Fish and Wildlife Service's classification hierarchy for wetlands and deepwater habitats of the United States | 13 |
| Figure 3. Distribution of wetlands in the conterminous United States | 22 |
| Figure 4. Physiographic regions used for regional analysis of national wetland-trends data | 23 |
| Figure 5. Model of human-induced impacts on wetlands, including effects on water level, nutrient status, and natural disturbance | 29 |
| Figure 6. Conceptual model of hydrology and its relationship to other components in a wetland ecosystem. | 34 |
| Figure 7. Nitrogen transformations in wetlands | 37 |
| Figure 8. Nutritional-resource categories and invertebrate-functional-feeding-group categories in lentic ecosystems | 41 |
| Figure 9. Compositional, structural, and functional biodiversity, shown as interconnected spheres, each encompassing multiple levels of organization | 52 |
| Figure 10. Categories of species that are classified and included in biodiversity | 60 |

| | | |
|------------|---|----|
| Figure 11. | Summary of processes leading to high diversity in terrestrial ecosystems | 65 |
| Figure 12. | Mallard food use and habitat preferences during winter in upper portions of the Mississippi Alluvial Valley | 76 |
| Figure 13. | Flooding regimes suggested for southern greentree reservoirs | 78 |

List of Tables

| | | |
|-----------|--|----|
| Table 1. | Early Wetland Classification by U.S. Fish and Wildlife Service | 11 |
| Table 2. | Hypothetical Example of a Wetland Profile Based on the Hydrogeomorphic Classification Scheme | 17 |
| Table 3. | Wetland Types of Mitsch and Gosselink and Their Equivalents in the National Wetlands Inventory | 18 |
| Table 4. | Natural and Human-Related Characteristics of Omernik's Aggregations of Ecoregions of the Conterminous United States | 25 |
| Table 5. | Types of Alterations to Wetlands | 30 |
| Table 6. | Comparison of Mineral and Organic Soils in Wetlands | 36 |
| Table 7. | Plant Life-Forms in Freshwater Wetlands of the United States | 39 |
| Table 8. | Characteristics of High-Nutrient and Low-Nutrient Wetlands | 42 |
| Table 9. | Functions of Wetlands and Their Value | 46 |
| Table 10. | Indicator Variables and Tools/Techniques for Inventorying, Monitoring, and Assessing Terrestrial Biodiversity at Four Levels of Organization, Including Compositional, Structural, and Functional Components | 53 |
| Table 11. | Examples of Benefits from Ecosystem, Species, and Genetic Diversity | 59 |
| Table 12. | Wetland-Dependent Taxa Federally Listed as Threatened or Endangered | 62 |

| | | |
|-----------|--|----|
| Table 13. | Number of Threatened and Endangered Species by Wetland-Habitat Type | 63 |
| Table 14. | Wetland Attributes that Assist in the Maintenance of Biodiversity | 64 |
| Table 15. | Habitat Types and Major Wildlife Taxa of Southern- Floodplain Forests | 70 |
| Table 16. | Habitat Types and Major Wildlife Taxa of Glacial Wetlands | 71 |
| Table 17. | Benefits and Problems Associated with Greentree Reservoir Management and Operation | 77 |
| Table 18. | Major Federal Laws, Directives, Regulations, and Programs Used for the Management, Protection, and Restoration of Wetlands | 90 |
| Table 19. | States That Have Comprehensive Wetland Laws for Inland Waters | 92 |

Preface

The work described in this report was authorized by Headquarters, U.S. Army Corps of Engineers (HQUSACE), as part of the Stewardship and Management Task Area of the Wetlands Research Program (WRP). The work was performed under Work Unit 32765, "Technology for Managing Wetlands," for which Mr. Chester O. Martin was the Technical Manager. Mr. Joe Wilson, CECW-OD, was the WRP Technical Monitor for this work.

Mr. Dave Mathis (CERD-C) was the WRP Coordinator at the Directorate of Research and Development, HQUSACE; Dr. William L. Klesch (CECW-PO) served as the WRP Technical Monitor's Representative; Dr. Russell F. Theriot, U.S. Army Engineer Waterways Experiment Station (WES), was the Wetlands Program Manager. Mr. Martin was the Task Area Manager.

This report was prepared by Mr. John H. Giudice and Dr. John T. Ratti at the University of Idaho, Moscow, as part of U.S. Fish and Wildlife Service (USFWS) Unit Cooperative Agreement No. 14-16-0009-1579, Work Order No. 11. This work was a cooperative effort among HQUSACE, USFWS, and the Office of Environmental Affairs, National Resource Damage Assessment, Department of Interior. Mr. Martin was the Project Manager under the general supervision of Mr. Hollis H. Allen, Chief, Stewardship Branch, and Dr. Robert M. Engler, Chief, Natural Resources Division, Environmental Laboratory (EL), WES. Dr. Edwin A. Theriot was the Assistant Director, EL, and Dr. John W. Keeley was Director, EL.

Review and comments were provided by Dr. Kent C. Jensen and Mr. Dan Smith, EL, WES, and Dr. Mark Konikoff, University of Southwestern Louisiana.

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1 Introduction

Interior, freshwater wetlands of the United States include potholes, marshes, swamps, bogs, fens, and riparian wetlands. These wetlands are diverse in form and function and have unique properties that make them different from both terrestrial and aquatic ecosystems (Figure 1). The biological complexity and ecological functions of wetlands make these ecosystems among the most valuable and productive on earth (Mitsch and Gosselink 1986:11). For example, wetlands are valued for functions such as surface-water storage, groundwater recharge, removal and transformation of nutrients, and soil stabilization (Greeson, Clark, and Clark 1979; Smith et al., In Preparation). Wetlands also provide habitat for fish and wildlife, including a third of the nation's threatened and endangered species (National Research Council (NRC) 1992a:265).

Despite their ecological importance, wetlands have been lost and degraded at alarming rates. An estimated 53 percent of the original 89.4 million ha of wetlands in the lower 48 States were lost by the mid-1970s (Dahl 1990), and losses continued at approximately 117,000 ha per year from the mid-1970s to the mid-1980s (Dahl and Johnson 1991). The national policy has changed gradually over the past 20 years from "encouraging the draining and filling of 'worthless swampland' to recognizing the many benefits provided by wetlands and considering them a valuable national resource" (Ratti and Kadlec 1992:1). Legislation (Federal, State, and local) and conservation programs have been enacted that directly or indirectly protect wetland ecosystems, e.g., Section 404 of the Federal Water Pollution Control Act (later amended as the Clean Water Act of 1977), the North American Wetlands Conservation Act, the Foods Securities Act (Swampbuster), the Wetland Reserve Program, and the Water Bank Program. Nevertheless, wetland loss and degradation continue.

Most wetland losses and degradation are the result of human population growth, technology, and increased resource demands. This trend is not likely to change in the near future (Ehrlich and Ehrlich 1990). Consequently, protection and preservation programs by themselves are not an adequate strategy for conserving wetlands and their functions. Supplemental strategies such as wetland restoration and creation are needed.

The science of habitat restoration/creation offers exciting possibilities because of the extensive nature of degraded ecosystems (Jordan 1988). Restoration and creation techniques are also being used to mitigate wetland losses and perturbations. Although this is a step in the right direction, there is much to learn about the functional success of such projects; that is, restored or created

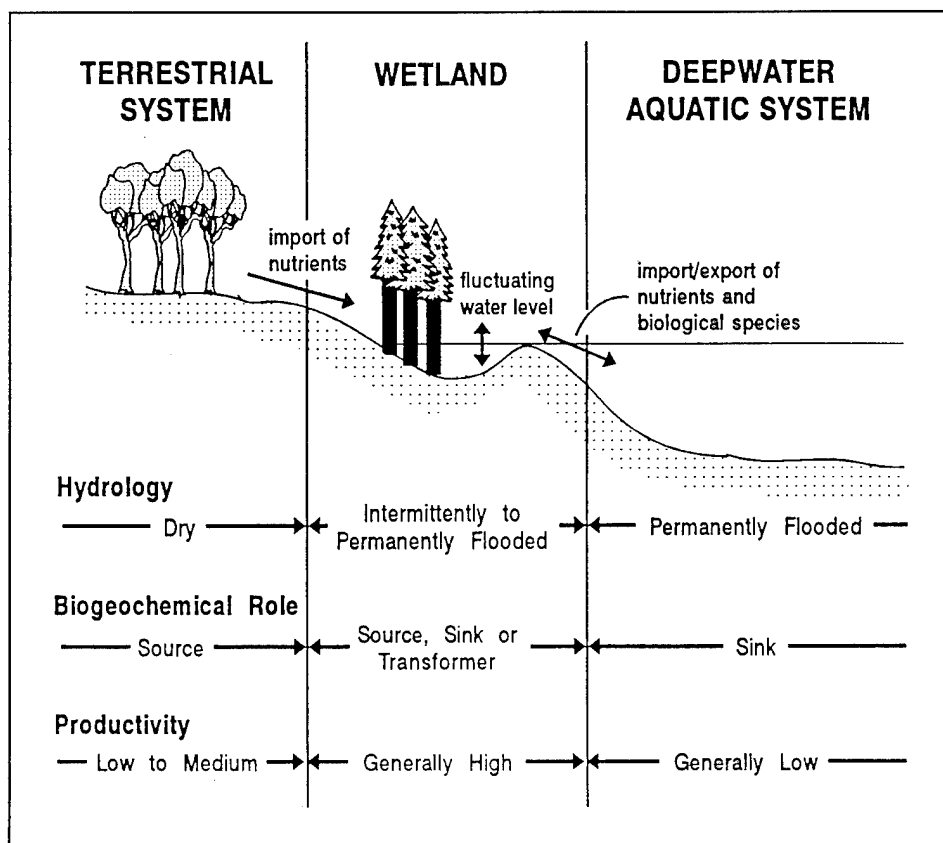


Figure 1. Wetlands are often located at the ecotones between terrestrial systems and permanently flooded deepwater-aquatic systems such as rivers, lakes, estuaries, or oceans. As such, they have an intermediate hydrology, a biogeochemical role as source sink or transformer of chemicals, and generally high productivity if they are open to hydrologic and chemical fluxes (from Mitsch and Gosselink (1993:17); copyright 1993 by Van Nostrand Reinhold, reprinted with permission)

wetlands may look natural, but there are few data to demonstrate that they function like natural, undisturbed systems (Zedler 1988; D'Avanzo 1990; Weller 1990; Zedler and Weller 1990). Furthermore, the emphasis of wetland-restoration programs is usually on the number of sites or total area restored or created. Little information exists on juxtaposition, wetland type, and hydrology of lost wetlands. Consequently, the type of wetlands being restored and created may be dissimilar from those being lost (Laubhan and Fredrickson 1993).

No single-wetland type provides the resources required by all species in a given period, nor does a single-wetland type provide the resources required for all stages in the annual cycle of a single species (Swanson, Kraper, and Serie 1979; Weller 1982; Fredrickson and Reid 1986; Weller 1990; Fredrickson and Batema 1992; Reid 1993; Laubhan and Fredrickson 1993). Consequently, loss of wetland heterogeneity may have important implications for efforts to restore and maintain the unique biological diversity (commonly termed

biodiversity) of a region. Managers and policymakers should strive to recreate and maintain a mosaic of wetland habitats that mirrors the unique diversity of the historic system, if possible (Parcells and Dunstan 1993). This can best be accomplished through a combined strategy of wetland protection, restoration, creation, and management (Ratti and Kadlec 1992; Laubhan and Fredrickson 1993).

Ecosystem management and the conservation of biodiversity has become an important public-policy issue in the United States. Although national attention has focused on terrestrial ecosystems and species, wetland management is also being affected by these issues. For example, wetland managers are being asked to place more emphasis on biodiversity and natural-community characteristics, while simultaneously maintaining more traditional wetland functions and values (Laubhan and Fredrickson 1993). To effectively meet this challenge, wetland managers will need to understand the principles and concepts of conservation biology and ecosystem management and be able to integrate this knowledge with information on the ecology, management, and biopolitics of freshwater wetlands.

This report is intended to (a) provide field and District-level personnel with an overview of the principles, concepts, strategies, and techniques necessary to preserve, restore, create, and manage natural-community and biodiversity characteristics of nontidal freshwater wetlands of the United States (excluding Hawaii) and (b) direct interested personnel to more detailed sources of information on selected topics related to wetland management and conservation biology. The report is designed to be used as a primer; thus, it is not intended to duplicate or displace more comprehensive and detailed reviews on wetland ecology and management (e.g., Good, Whigham, and Simpson 1978; Brinson et al. 1981; Mitsch and Gosselink 1986; Weller 1987; Hook et al. 1988a, b; van der Valk 1989; Niering 1985, 1991; Payne 1992; Fredrickson and Batema 1992). Readers are urged to consult these and other identified sources for more detailed information prior to initiating wetland management, restoration, or creation projects.

Because of time and space constraints, only select references were cited in this review. However, a more comprehensive bibliography has been developed as part of the overall project. The bibliography contains citations on ecology, management, and biodiversity of interior, freshwater wetlands of the United States. It was compiled using Pro-Cite software (Personal Bibliographic Software, Inc.), a popular literature-management program. The program has keyword-searching capabilities and permits easy formatting of citations, including downloading to ASCII files in mail-merge format for use in other database programs. Data fields include Author, Date of Publication, Title, Source, and Keywords.

2 Defining Wetlands

Many wetlands are transitional ecosystems; that is, they exist in a half-way world between aquatic (deepwater) and terrestrial ecosystems, often exhibiting characteristics of each (Smith 1980:225; Gopal et al. 1990:9). However, there is no single, universally recognized definition that adequately describes all wetland types (Cowardin et al. 1979:3; Mitsch and Gosselink 1986:16). The problem of definition arises because (a) freshwater wetlands are highly diverse (ranging from temporarily flooded riparian areas to more permanently flooded deepwater swamps and marshes), (b) the demarcation between dry and wet environments lies along a continuum, and (c) the reasons or needs for defining wetlands vary among interest groups (Cowardin et al. 1979:3). For example, wetland scientists need a flexible but rigorous definition that can be used in classification, inventory, and research; wetland managers are more concerned with regulations governing wetland modification/protection and thus need clear, legally binding definitions; and policymakers need a definition that accommodates broad regional differences in wetlands and allows wetlands to be identified even in dry periods.

Wetland definitions frequently used by managers and scientists in the United States include the "Circular 39" definition, the current U.S. Fish and Wildlife Service (USFWS) definition, regulatory definitions arising from Section 404 of the Clean Water Act, and statutory definitions associated with legislation. These definitions are described in the following paragraphs. There also are several international (Gore 1983; Gopal et al. 1990:9) and Canadian definitions (Tarnocai 1979:11; Zoltai 1979:1), but they will not be discussed here.

Circular 39 Definition

Circular 39, a USFWS report, described the extent and value of wetlands for waterfowl and other wildlife in the United States (Shaw and Fredine 1956). The report summarized results of the first national wetlands inventory, which was based on the wetland classification of Martin et al. (1953). In addition, the report contained a definition of wetlands that is still frequently used today (Shaw and Fredine 1956:3):

...lowlands covered with shallow and sometimes temporary or intermittent waters. They are referred to by such names as marshes, swamps, bogs, wet meadows, potholes, sloughs, and river-overflow

lands. Shallow lakes and ponds, usually with emergent vegetation as a conspicuous feature, are included in this definition, but the permanent waters of streams, reservoirs, and deep lakes are not included. Neither are water areas that are so temporary as to have little or no effect on the development of moist-soil vegetation.

Despite its limitations (i.e., broad-wetland categories and emphasis on waterfowl habitat), Circular 39 served the needs of both wetland managers and wetland scientists (Knighton 1985; Mitsch and Gosselink 1986:17). Furthermore, much of the wetland legislation passed in the United States refers to wetland categories and definitions described in Circular 39 (Payne 1992:419).

Current USFWS Definition

A more comprehensive definition was developed by USFWS wetland scientists and was presented in a report entitled "Classification of Wetlands and Deepwater Habitats of the United States" (Cowardin et al. 1979:3):

Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. ... wetlands must have one or more of the following attributes: (1) at least periodically, the land supports predominantly hydrophytes; (2) the substrate is predominantly undrained hydric soil; and (3) the substrate is nonsoil and is saturated with water or covered by shallow water at some time during the growing season of each year.

This definition, and the subsequent classification scheme, was the product of extensive peer review, comments from State and Federal agencies, field testing, and use during the early phases of the National Wetlands Inventory (NWI) (Cowardin 1982a). Although it is a broad, flexible, and comprehensive definition, it still excludes important habitats such as floodplains that are temporarily flooded during the nongrowing season. Furthermore, areas with drained hydric soils that are now incapable of supporting hydrophytes because of a change in water regime are not considered wetlands under this definition; however, these areas may be quite suitable for restoration (Cowardin et al. 1979:3). Despite these shortcomings, the current USFWS definition has been widely accepted by wetland scientists and has allowed more detailed wetland classification and inventory than was possible with the Circular 39 definition. However, its primary utility is with scientific studies and habitat inventory; generally, it is less suited to management or regulation of wetlands (Mitsch and Gosselink 1986:18).

Regulatory and Statutory Definitions

Probably the most widely accepted regulatory definition of wetlands is the one used by the U.S. Army Corps of Engineers (USACE) and the U.S. Environmental Protection Agency (USEPA):

... those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands include swamps, marshes, bogs, and similar areas. (USEPA, 40 CFR 230.3, December 24, 1980; USACE, 33 CFR 328.3, November 13, 1986)

A similar definition exists in the Swampbuster legislation that governs the Department of Agriculture and in the Emergency Wetlands Resources Act of 1986. However, these statutory definitions explicitly require the presence of hydric soils, a condition that is only implicit in the USEPA's and USACE's regulatory definition. In addition, the Emergency Wetlands Restoration Act provides congressional definitions of hydric soil and hydrophytic vegetation (16 U.S.C. Secs. 3801[a][16];3902), and the Swampbuster definition excludes wetlands that were converted to cropland prior to 1985.

Defining wetlands and delineating their boundaries based on the presence of water and characteristics of the soil and vegetation can be complex because (a) many wetlands are seasonally dry each year (maybe even for years during a drought), and (b) many species of plants that exist in wetlands are also common outside wetlands (Environmental Defense Fund and World Wildlife Fund 1992:10). Moreover, in a regulatory sense, the relative "wetness" of a site does not necessarily indicate its value. For example, temporary and seasonal wetlands may appear to be insignificant and can easily be drained or altered; however, they can play an important role in water quality protection and the maintenance of biodiversity.

No single definition will meet the needs and desires of all interested parties. Consequently, legal definitions of wetlands, and thus jurisdictional protection, continue to be debated in court, by Congress, and among the Federal and State agencies responsible for wetland protection (see Committee on Science, Space, and Technology 1991; Environmental Defense Fund and World Wildlife Fund 1992). This issue is not likely to be resolved until the National Academy of Sciences completes its study on the science of wetland definition and delineation. Their report and recommendations were due in 1994.

Riparian Ecosystems

Recently, agency and public attention has focused on management and restoration of riparian ecosystems, especially in western states (Anderson 1987). Traditionally, riparian ecosystems are thought of as being associated with riverine systems. For example, Jensen and Platts (1990:368) described riparian as being transitional between aquatic (river or stream channel) and upland habitat. However, Johnson, Carothers, and Simpson (1984) argued for a broader definition of riparian ecosystems:

...on or pertaining to land adjacent to riverine and estuarine channels, lacustrine beds, or oases and other sites where surface water and/or

groundwater occurs in excess of on-site precipitation; occupied by biotic communities differing in species composition and/or population densities from those of the surrounding uplands due to the substrate: a) being periodically covered with water; b) having higher soil moisture; or c) in the case of rocky banks or cliffs, existing plant and animal species are dependent on a proximity to water.

Although the terms "riparian" and "wetlands" are used interchangeably in certain cases, the concepts are not necessarily coextensive (Ratti and Kadlec 1992:43). For example, mesic or xeric (infrequently wet) sites along the riparian continuum do not meet the legal or biological definition of a wetland; however, such areas may still be functionally unique as compared with the adjacent upland.

3 Wetland Classification

Wetland scientists have devised different schemes to classify wetlands and inventory their extent, distribution, and function. These classification schemes are valuable tools for both wetland scientists and managers, but they are only valuable if the user is familiar with their scope and limitations (Mitsch and Gosselink 1986:450). For example, any classification scheme involves artificially dividing up what is really an ecological continuum. Hence, some wetlands may seem to fit into more than one category and may be judged differently by different investigators. This problem is exacerbated by the dynamic nature of most wetlands. Depending on the classification criteria used, a single-wetland can functionally span several categories within a few years of time (Weller 1987:15). These temporal changes are important because wetland types and habitats differ in their attractiveness to wildlife species (Weller and Spatcher 1965; Burger 1985; Fredrickson and Reid 1986) and because wetland distribution varies regionally as do animal associations (Weller 1987:15).

Conceptual and Semantic Problems

The process of classification and results of inventories depend on the conceptual framework of the classification scheme. Cowardin (1982b:58) cautioned that there are two different concepts for the elements (i.e., wetland habitats) being classified:

Area concept—the element of classification is viewed as an area of the earth's surface that is homogeneous for a set of attributes described in the classification. The majority of classifications of this type use some combination of hydrologic, edaphic, and biotic attributes for defining classes.

Physiognomic concept—the element classified is a physical entity on the landscape such as a pond, lake, estuary, or segment of a river. The area within the unit is seldom homogeneous for hydrologic, edaphic, and biotic attributes.

To illustrate the difference, consider a wetland basin containing a central expanse of submergent vegetation, a peripheral band of cattail,¹ and an outer band of marsh grasses and sedges. Under the area concept, this basin would be classified into three wetland types, each representing a relatively homogeneous community. In contrast, under the physiognomic concept, the entire heterogeneous basin would be classified as one wetland type based on some criteria (e.g., dominant vegetation found in the deepest zone of the wetland). Both concepts have important applications to management and research (see Cowardin 1982b); however, care must be taken when comparing results of classifications based on different concepts. Furthermore, semantic problems can add to the confusion created by conceptual differences. For example, terms such as wetland, wetland basin, and pond may have different meanings in different classification schemes. Wetland classification depends on well-understood definitions; thus, terms should be defined before making comparisons among classification schemes.

Classification Schemes

There are numerous wetland-classification schemes; however, most classifications were designed for a specific geographic area or a restricted range of wetland types (Brinson 1993). For example, classification schemes have been published for prairie wetlands (Stewart and Kantrud 1971; Millar 1976), wetlands of South Dakota (Evans and Black 1956), wetlands of glaciated regions (Golet and Larson 1974; Hollands 1987), coastal wetlands (Odum, Copeland, and McMahan 1974), forested wetlands of Florida (Lugo and Snedaker 1974; Wharton et al. 1976), and peatlands of Minnesota (Heinselman 1963, 1970). A few wetland-classification schemes have been designed for broad-scope coverage (e.g., Martin et al. 1953; Gosselink and Turner 1978; Cowardin et al. 1979; Brinson 1993); however, each scheme has different goals, objectives, and classification criteria.

Several authors have reviewed wetland classification and the various classification schemes (see Stewart and Kantrud 1971; Cowardin et al. 1979; Hofstetter 1983; Mitsch and Gosselink 1986; Mader 1991; Payne 1992; and Brinson 1993). It was not the intention of the authors of this report to duplicate these more comprehensive reviews. Instead, the focus was on three wetland-classification schemes that have broad coverage (geographic and range of wetland types) and that have influenced management and regulatory decisions in the United States. Two schemes (Martin et al. 1953; Cowardin et al. 1979) emphasize biotic characteristics of wetlands and were designed for national wetland inventories. The third scheme (Brinson 1993) emphasizes abiotic features and was designed to support ongoing efforts to develop methods for assessing physical, chemical, and biological functions of wetlands. Also provided is a report on a simple classification scheme (Mitsch and Gosselink 1986:24) designed to facilitate discussions of wetland ecology, management, and biodiversity.

¹ See Appendix A for scientific names of plants named in text.

Martin et al. classification

The scheme of Martin et al. (1953) was the most widely used wetland classification in the United States prior to 1979. It was developed for the first national wetlands inventory, and the results of both the classification scheme and the inventory were published in Circular 39 (Shaw and Fredine 1956). The scheme divided wetlands into 20 types under four major categories (Table 1). Wetland types were based on criteria such as water depth and permanence, salinity, vegetative life form, and dominant-plant species. The classification is simple but has been criticized for having categories that are too broad with inadequate descriptions for detailed differentiation of somewhat similar types (Leitch 1966; Stewart and Kantrud 1971; Golet and Larson 1974; Cowardin et al. 1979:2). Nevertheless, the basic scheme influenced other classification efforts such as Evans and Black (1956), Stewart and Kantrud (1971), Golet and Larson (1974), and Goodwin and Niering (1975). Although the USFWS officially adopted a new classification scheme in 1979 (Cowardin et al. 1979), wetland scientists and managers still frequently refer to Circular 39 because it is the most widely known wetland-classification scheme with wildlife management emphasis (Knighton 1985).

Cowardin et al. classification

In 1974, the Office of Biological Survey of the USFWS began a new national inventory of wetlands (Cowardin et al. 1979:2). Because of the narrow focus and weaknesses inherent in Circular 39, and because wetland ecology had become significantly better understood since 1954, the USFWS elected to design a new wetland-classification scheme (Cowardin et al. 1979:2). Designers of the new classification, "Classification of Wetlands and Deepwater Habitats of the United States," had four long-range objectives: (a) to describe ecological units that would have certain homogeneous natural attributes; (b) to arrange these units in a scheme that would aid decisions about resource management; (c) to furnish units for inventory and mapping; and (d) to provide uniformity in concepts and terminology. Because wetlands were defined as being continuous with deepwater ecosystems, both categories were included in the new classification. Thus, the Cowardin et al. classification was designed to include nearly all aquatic and semiaquatic ecosystems located in the conterminous United States.

The classification is hierarchical, progressing from systems and subsystems (the most general levels) to classes, subclasses, and dominance types (Figure 2). Wetland and deepwater habitats are grouped according to hydrologic, geomorphic, chemical, and biological factors. Cowardin et al. (1979:4-12) defined and described the limits of each system:

- a. *Marine*—open ocean overlying the continental shelf and its associated high-energy coastline.

Table 1
Early Wetland Classification by U.S. Fish and Wildlife Service¹

| Type | Descriptor | Site Characteristics |
|---|------------------------------------|--|
| Inland Fresh Areas | | |
| 1 | Seasonally flooded basins or flats | Soil in upland depressions and bottomlands covered with water or waterlogged during variable periods, but well drained during much of the growing season, with bottomland hardwoods and herbaceous plants. |
| 2 | Fresh meadows | Waterlogged to within a few centimeters of surface, but without standing water during growing season; herbaceous plants. |
| 3 | Shallow fresh marshes | Soil waterlogged and often covered with ≥ 15 cm of water; emergents during growing season. |
| 4 | Deep fresh marshes | Soil covered with ≥ 15 cm to 0.9 m of water during growing season; submergents. |
| 5 | Open fresh water | Water < 3 m deep; submergents, fringed with emergents. |
| 6 | Shrub swamps | Soil waterlogged during growing season, often covered with ≥ 15 cm of water; swamp shrubs. |
| 7 | Wooded swamps | Soil waterlogged; spongy covering of mosses, with other herbaceous and woody plants. |
| 8 | Bogs | Soil waterlogged; spongy covering of mosses, with other herbaceous and woody plants. |
| Inland Saline Areas | | |
| 9 | Saline flats | Flooded after periods of heavy precipitation; waterlogged within a few centimeters of surface during growing season; salt-tolerant herbs. |
| 10 | Saline marshes | Soil waterlogged during growing season; often covered with 0.7 to 1 m of water; shallow lake basins; alkali or hardstem bulrush, sago, and widgeon grass. |
| 11 | Open saline water | Permanent areas of shallow saline water of variable depth; submergents. |
| ¹ Source: Payne (1992:422), after Shaw and Fredine (1956). | | |
| (Continued) | | |

| Table 1 (Concluded) | | |
|-----------------------------|----------------------------------|---|
| Type | Descriptor | Site Characteristics |
| Coastal Fresh Areas | | |
| 12 | Shallow fresh marshes | Soil waterlogged during growing season; at high tide ≥ 15 cm of water; on landward side, deep marshes along tidal rivers, sounds, deltas; grasses and emergents. |
| 13 | Deep fresh marshes | At high tide covered with 15 cm to 0.9 m of water during growing season; along tidal rivers and bays; emergents and often submergents. |
| 14 | Open fresh water | Shallow portions of open water along fresh tidal rivers and sounds; plants absent or emergents in water < 1.8 m. |
| Coastal Saline Areas | | |
| 15 | Salt flats | Soil waterlogged during growing season; sites occasionally to fairly regularly covered by high tide; landward sides or islands within salt meadows and marshes; sparse grasses. |
| 16 | Salt meadows | Soil waterlogged during growing season; rarely covered by tide water; landward side of salt marshes; grasses and sedges. |
| 17 | Irregularly flooded salt marshes | Covered by wind tides at irregular intervals during growing season; along shores of nearly enclosed bays, sounds, etc.; needlerush. |
| 18 | Regularly flooded salt marshes | Covered at average high tide with ≥ 15 cm of water; along open ocean and sounds; salt-marsh cordgrass on Atlantic, alkali bulrush on Pacific. |
| 19 | Sounds and bays | Portions of saltwater sounds and bays shallow enough to be diked and filled; all water landward from average low-tide line; submergents. |
| 20 | Mangrove swamps | Soil covered at average high tide with 15 cm to 0.9 m of water; along coast of southern Florida; mangroves. |

- b. Estuarine*—deepwater tidal habitats and adjacent tidal wetlands that are usually semi-enclosed by land but have open, partly obstructed, or sporadic access to the open ocean, and in which ocean water is at least occasionally diluted by freshwater runoff from the land.
- c. Riverine*—all wetlands and deepwater habitats contained within a channel, with two exceptions: (1) wetlands dominated by trees, shrubs, persistent emergents, emergent mosses, or lichens, and (2) habitats with water containing ocean-derived salts in excess of 0.5 parts per thousand.

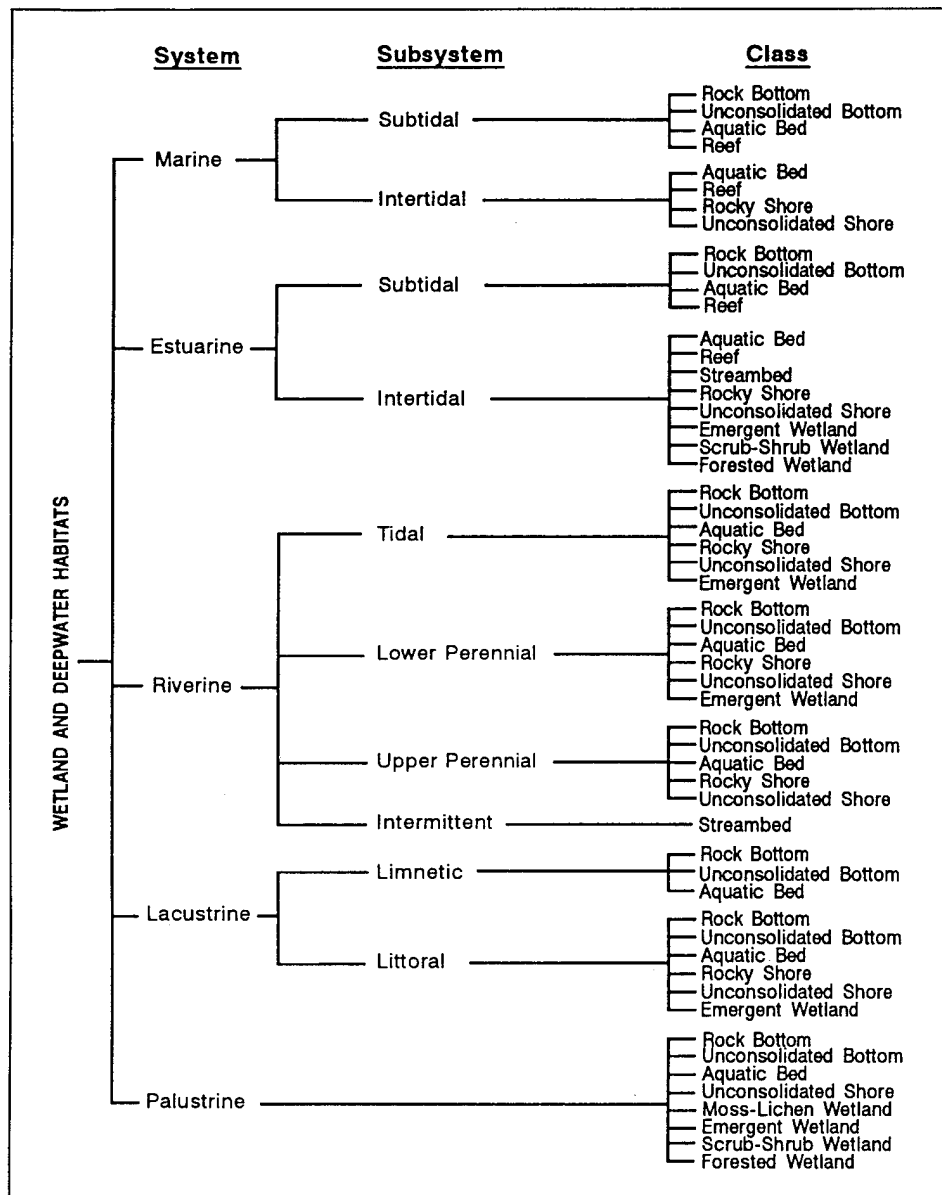


Figure 2. U.S. Fish and Wildlife Service's classification hierarchy for wetlands and deepwater habitats of the United States (from Cowardin et al. 1979:5)

- d. *Lacustrine*—wetlands and deepwater habitats with all of the following characteristics: (1) situated in a topographic depression or a dammed river channel; (2) lacking trees, shrubs, persistent emergents, emergent mosses or lichens with greater than 30% areal coverage; and (3) total area exceeds 8 ha. Similar wetland and deepwater habitats totaling less than 8 ha are also included in the *Lacustrine* system if an active wave-formed or bedrock shoreline feature makes up all or part of the boundary, or if the water depth in the deepest part of the basin exceeds

2 m at low water. Lacustrine waters may be tidal or nontidal, but ocean-derived salinity is always less than 0.5 ppt.

- e. *Palustrine*—all nontidal wetlands dominated by trees, shrubs, persistent emergents, emergent mosses or lichens, and all such wetlands that occur in tidal areas where salinity due to ocean-derived salts is below 0.5 ppt. It also includes wetlands lacking such vegetation, but with all of the following four characteristics: (1) area less than 8 ha; active wave-formed or bedrock shoreline features lacking; (3) water depth in the deepest part of basin less than 2 m at low water; and (4) salinity due to ocean-derived salts less than 0.5 ppt.

Two meters was used as the lower limit for inland wetlands because this represents the maximum depth that emergent plants normally grow (Sculthorpe 1967). If plants are growing in water >2 m deep, then the boundary between wetland and deepwater habitats goes to the deepwater edge of the zone of emergent plants. The Riverine and Lacustrine systems include both deepwater and wetland habitats, whereas the Palustrine system includes only wetland habitats. However, palustrine wetlands can be associated with Riverine and Lacustrine systems. For example, “palustrine wetlands may be situated shoreward of lakes, river channels, or estuaries; on river floodplains; in isolated catchments; or on slopes. They may also occur as islands in lakes or rivers” (Cowardin et al. 1979:10). Palustrine wetlands are most germane to this report because (a) they support a wide diversity of plants and animals (including species associated with uplands), (b) most strategies and techniques for managing freshwater-wetland wildlife were developed for palustrine wetlands (i.e., shallow, vegetated wetlands), and (c) palustrine wetlands are freshwater systems (with the exception of salt and brackish marshes of arid and semiarid regions) prevalent in noncoastal or interior areas, which are the focus of this report.

There are eight classes of palustrine wetlands, but most fall into one of three types: (a) emergent, (b) scrub-shrub, and (c) forested. Emergent wetlands are often dominated by herbaceous vegetation such as rushes, sedges, grasses, cattails, arrowheads, pondweeds, and water lilies. These wetlands are commonly referred to as marsh, meadow, fen, prairie pothole, and slough. Scrub-Shrub wetlands are dominated by woody vegetation <6 m tall and include shrub swamps, shrub-carr habitat, bogs, and pocosins. Forested wetlands are dominated by woody vegetation (trees) >6 m tall and include spruce/larch bogs, cedar/maple swamps, and bottomland-hardwood forests.

Although the Cowardin et al. classification has broad-wetland coverage, it has been criticized for excluding riparian habitats, which are some of the most unappreciated and abused wetland areas of the United States (Johnson, Carothers, and Simpson 1984). Johnson, Carothers, and Simpson (1984) proposed a riparian-classification scheme that would add three subsystems to the Palustrine system of Cowardin et al. (1979):

- a. *Hydroriparian*: Wetlands with hydric soils or whose substrates are never dry or are dry for only a short period; usually associated with

perennial or intermittent water. Vegetation, when present, consists of predominance of obligate and preferential wet riparian plants.

- b. *Mesoriparian*: Wetlands with nonhydric soils and whose substrate is dry seasonally; usually associated with intermittent water or high-elevation ephemeral wetlands. Vegetation, when present, consists of a mixture of obligate, preferential, and facultative riparian plants.
- c. *Xeroriparian*: Mesic to xeric-habitat type with average annual moisture higher than surrounding uplands, but provided with surface moisture in excess of local rainfall only on infrequent occasions (usually for less than 1 month per year). Vegetation, when present, consists of a mixture of preferential, facultative, and nonriparian plants.

Johnson, Carothers, and Simpson (1984) described these subsystems in more detail and discussed possible delineation criteria (i.e., indicator assignments of plant species based on their relationship to the riparian zone).

Brinson's hydrogeomorphic classification

Wetland classifications such as Martin et al. (1953) and Cowardin et al. (1979) placed great emphasis on the structure and species composition of the plant community, which was necessary to meet their major goal of wetland inventory and monitoring. In contrast, hydrogeomorphic classification (Brinson 1993) places emphasis on abiotic features (i.e., hydrologic and geomorphic controls) that are thought to be responsible for maintaining many wetland functions. Wetland functions are processes necessary for self-maintenance of the ecosystem (e.g., primary production, nutrient cycling, and decomposition) and should not be confused with wetland values. As explained in Brinson (1993:A5), the term "values" is associated with society's perception of ecosystem functions, whereas functions occur in ecosystems regardless of whether or not they have values. In other words, "functions exist in the absence of society and are normally part of the self-sustaining properties of an ecosystem" (Brinson 1993:3).

The need for a functionally based classification scheme is twofold (Brinson 1993:12-13):

- a. ...to simplify our concept of wetlands, recognizing that while each one may be unique, each can be placed into categories in which similar wetlands share functional properties. The result of this simplification should be improved communication among researchers and managers, and perhaps even with the public, by focusing on processes that are fundamental to the sustained existence of these ecosystems.
- b. ...to foster the development and the redevelopment of paradigms that clarify the relationship between ecosystem structure and function.

Although abiotic characteristics are emphasized in this classification scheme, it is recognized that biotic factors also play important roles in the structure and function of wetlands. Hence, "familiarity with the adaptations and tolerance limits of plant and animal species is necessary skill for successful classification within a given biogeographic region" (Brinson 1993:2).

The hydrogeomorphic-classification scheme is based on three properties: (a) geomorphic setting (i.e., topographic location of a wetland within the surrounding landscape), (b) water source and its transport, and (c) hydrodynamics (i.e., direction of flow and strength of water movement within a wetland). These properties are further subdivided into categories, some of which are not mutually exclusive (e.g., water sources):

Geomorphic setting.

- a. Depressional wetlands (e.g., kettles, potholes, and vernal pools).
- b. Extensive peatlands (e.g., blanket bogs and tussock tundra).
- c. Riverine wetlands.
- d. Fringe wetlands (i.e., wetlands that occur in estuaries where tidal forces dominate or in lakes where water moves in and out of the wetland from the effects of wind, waves, and seiches).

Water sources (hydrologic inputs).

- a. Precipitation.
- b. Groundwater discharge (inflow, usually into and through wetland sediments).
- c. Surface or near-surface inflow (depending on the wetland, this could include flooding from tides, overbank flow from stream channels, and interflow or overland flow).

Hydrodynamic settings.

- a. Vertical fluctuations of the water table that result from evapotranspiration and subsequent replacement by precipitation or groundwater discharge.
- b. Unidirectional flows that range from strong channel-contained currents to sluggish sheet flow across a floodplain.
- c. Bidirectional flows (i.e., surface or near-surface flows resulting from tides or seiches).

Wetlands are described according to each property and category, indicators of function are recorded (e.g., high-water marks, soil texture, and species composition of the plant community) or derived from other data sources (e.g., maps and water quality data), and the ecological significance of each of the properties is determined. This information is used to develop wetland profiles (Table 2), which help reveal the functions that wetlands are likely to perform. Eventually, wetland profiles should lead to a population of reference wetlands. Reference wetlands represent benchmarks upon which other wetlands could be

| Table 2 Hypothetical Example of a Wetland Profile Based on the Hydrogeomorphic Classification Scheme ¹ | | | | |
|--|---|---|---|--|
| Examples of Properties | Qualitative Evidence | Quantitative Evidence | Functions | Ecological Significance |
| Geomorphic Setting: Depressional Wetland | | | | |
| Located in marginally dry climate (e.g., prairie pothole region). Variable inlets and outlets. | Inlets and outlets may be defined by contours or intermittent stream symbol. | If water has low conductivity, wetland is recharging underlying aquifer. If high conductivity, groundwater is discharging to wetland. | Retains inflow; loss primarily by evapotranspiration (ET) or infiltration. May be subject to wide fluctuation in water depth. | Geographic location critical to migrating waterfowl as flyway position indicates. Changes in vegetation create varied waterfowl habitat. May be vulnerable to eutrophication and toxin accumulation because of long residence time of water. Probable import and export of detritus. |
| Water Sources: All Three Sources (Precipitation, Groundwater Discharge, and Lateral Surface or Near-surface Flow) | | | | |
| All three sources of hydrologic input, but precipitation is minor (subhumid to semiarid). | Alternate drought and wet periods produce decade-long cycles of water table fluctuations. | Precipitation < PET (potential evapotranspiration) | High water levels induced by precipitation; groundwater (GW) discharge prevents extreme drawdowns; wetland may recharge GW when water table is high; conserves/reduces GW discharge when water levels are normal. | High primary production occurs when water is abundant; decomposition is rapid enough during drying periods to prevent peat accumulation. |
| Hydrodynamic Setting: Vertical Fluctuation of Water Table | | | | |
| Seasonal fluctuations nested within multiyear cycles. | Prairie pothole region. Soils diagnostic of dominant water sources. | Aerial photos show variable year-to-year extent of flooding. Hydrographs of water table confirm both short- and long-term fluctuations. | Landscape a mosaic of ponds varying in depth at a single point in time. Floodwaters retained by depressions. | Flyway and breeding sites for waterfowl. Retention of water results in aquatic/moist habitat in otherwise semiarid conditions. |
| ¹ Source: after Brinson (1993). | | | | |

compared for various purposes such as assessment, training (i.e., classification and functional interpretation), and mitigation.

Strengths of the hydrogeomorphic classification include its ability to relate hydrology and geomorphology to wetland function. Furthermore, the classification is open-ended (i.e., it does not have a finite number of discrete classes that are chosen a priori), which permits adaptations to various types of wetlands and geographic regions. However, more distinct classes of wetlands should emerge as the classification is applied and community profiles are developed in different physiographic regions of the country (Brinson 1993).

Mitsch and Gosselink

Although useful for inventories and scientific studies, many classification schemes are too detailed and complex to be used as a basis for discussing wetland ecology, restoration, biodiversity, and management. Mitsch and Gosselink (1986:24) suggested a simpler scheme for such purposes. They divided wetlands into four types of inland-wetland ecosystems and three types of coastal-wetland ecosystems (Table 3). This elementary classification is appealing and useful because the divisions (a) cover most of the wetlands found in the United States, (b) are generally recognized as distinct in form and function, and (c) are commonly distinguished in the literature (Payne 1992:423). Moreover, the use of these divisions allows a cohesive discussion of biodiversity and management concepts, while still maintaining a relationship to more complicated classifications. For example, inland-wetland systems of Mitsch and Gosselink (1993) correspond closely to basic wetland types found in the Palustrine system of Cowardin et al. (1979) (Table 3).

Table 3
Wetland Types of Mitsch and Gosselink and Their Equivalents
in the National Wetlands Inventory¹

| Wetland Types Used by Mitsch and Gosselink | National Wetlands Inventory Equivalent² |
|---|---|
| Coastal Wetland Ecosystems | |
| Tidal Salt Marshes | Estuarine intertidal emergent, haline |
| Tidal Freshwater Marshes | Estuarine intertidal emergent, fresh |
| Mangrove Wetlands | Estuarine intertidal forested and shrub, haline |
| Inland Wetland Ecosystems | |
| Inland Freshwater Marshes | Palustrine emergent |
| Northern Peatlands | Palustrine moss-lichen |
| Southern Deepwater Swamps | Palustrine forested and scrub-shrub |
| Riparian Wetlands | Palustrine forested and scrub-shrub |

¹ Source: Mitsch and Gosselink (1993:34); copyright 1993 by Van Nostrand Reinhold, reprinted with permission.

² Cowardin et al. (1979).

4 Inventory and Status of Wetlands

Inventory and Monitoring

Inventories determine the extent of various types of wetlands in a given region, whereas monitoring programs measure change in or impacts to a given region or system over a period of time. Both concepts have application to wildlife habitat, including wetlands (Cooperrider, Boyd, and Stuart 1986). The kind of inventory employed depends on the specific needs of the user. For example, Frayer (1988) distinguished between "policy" and "management" inventories. Policy inventories are designed to cover large areas and produce information that is used to determine if management or policy is necessary. However, policy inventories do not produce the detailed information necessary to manage a specific wetland or complex of wetlands. Conversely, a management inventory is intensive and provides site-specific information, although it may be expensive and time-consuming. Recent advances in mapping technology may make frequently updated national inventories (policy type) more site-specific; however, the current technology is expensive, and mapping accuracy is less than can be achieved with more common remote-sensing techniques such as aerial photography (Frayer 1988).

Remote sensing

Most large-scale wetland surveys use remote-sensing imagery to map or statistically sample wetlands. The imagery type employed depends on goals or objectives of the inventory (Montanari 1988:69) and on the resolution required, area to be covered, and cost of data collection (Mitsch and Gosselink 1986:469). The greatest amounts of wetland information can be derived from aerial photography (Roller 1977). Low-altitude-aircraft surveys are an effective way to survey small areas. Conversely, high-altitude-aircraft surveys offer much greater coverage and may be less expensive per unit area when costs of photo-interpretation are included (Mitsch and Gosselink 1986:469). Color and color-infrared photography are popular techniques for wetland inventories from aircraft (e.g., Cuplin 1978; Estrin 1986; Dahl and Johnson 1991), although black-and-white and black-and-white-infrared photography have been used with some success (e.g., Cowardin and Myers 1974; Roller and Colwell 1978; Dahl and Johnson 1991). High-resolution-multispectral-scanner

imagery from low-altitude-plane flights has also been used with some success (Mitsch and Gosselink 1986:471).

Satellite imagery such as LANDSAT has also been used to inventory and map wetlands (Anderson, Wentz, and Treadwell 1980:293-294), although the imagery is best suited to extensive, general-purpose wetland inventories (Roller 1977). Even large-scale inventories such as the National Wetlands Inventory may require more detail than LANDSAT can provide without additional data collected from aerial photographs and field work (Nyc and Brooks 1979 cited in Mitsch and Gosselink 1986:469). Furthermore, Werth and Meyer (1981) compared LANDSAT digital analysis with color-infrared photography for wetlands classification and concluded that LANDSAT digital analysis was faster but not as cost productive or accurate as aerial-photography methods. However, satellite imagery continues to improve. For example, SPOT MSS (a type of multispectral-scanner imagery) is becoming more widespread in wetland inventories and mapping because of its fine resolution. Even with current limitations, satellite imagery has many applications for wetland studies (e.g., LaPerriere and Morrow 1978; Gilmer, Colwell, and Work 1978; Best and Moore 1981; Jensen et al. 1984), and "there is no good reason for not utilizing both aerial photography and satellite imagery within the same inventory as long as the established goals and objectives are met in a cost-effective manner" (Montanari 1988:69).

The review of remote sensing was limited to a brief description of common techniques used in wetland inventory and monitoring. More comprehensive discussions have been published (Anderson, Wentz, and Treadwell 1980; Mitsch and Gosselink 1986; Montanari 1988), including an introduction to the principles and theory of remote sensing (Colwell 1983). Furthermore, Lampman (1993) recently prepared an extensive bibliography of remote sensing techniques used in wetland research.

National wetland inventories

The USFWS has conducted two nationwide surveys of wetlands in the United States. The first survey was started in 1954 (see Shaw and Fredine 1956). The second survey (the National Wetlands Inventory (NWI)) was initiated in 1974 (Gebhard 1988). Although the first survey covered approximately 40 percent of the conterminous United States (primarily in the Mississippi flyway), it focused on wetland areas important to waterfowl (Gebhard 1988). In contrast, the NWI had broader geographic coverage and included nearly all wetland and deepwater habitats. The Cowardin et al. (1979) classification scheme was developed to meet the needs of the NWI.

Map production from the NWI became operational in 1980. The USFWS is scheduled to complete mapping of the contiguous United States by 1998 as required by the Emergency Wetlands Resources Act of 1986. Wetland mapping of Alaska will be completed as soon as possible thereafter. By June of 1992, the USFWS had produced detailed maps covering 72 percent of the contiguous United States, 22 percent of Alaska, and all of Hawaii (USFWS

Newsletter NWI 6/92). Furthermore, the USFWS has computerized (digitized) more than 8,113 of its wetland maps representing 15 percent of the conterminous United States (USFWS Newsletter NWI 6/92). Gebhard (1988) briefly described this georeferenced database.

As part of the NWI, the USFWS designed and implemented the first comprehensive, statistically valid effort to estimate the Nation's wetlands (Fraye 1988). The USFWS's first report on status and trends of the Nation's wetlands (Fraye et al. 1983) estimated the rate of wetland conversion between the mid-1950s and the mid-1970s. Reports by Dahl (1990) and Dahl and Johnson (1991) described wetland status and trends from the mid-1970s to the mid-1980s. These reports did not address causes for changes in wetland acreage or effects of wetland loss on the Nation's fish and wildlife resources. However, Tiner (1984) described the use/value of wetlands, identified national problem areas, and made management recommendations based on the NWI. Scientists have also described status and trends of wetlands in specific States/regions (e.g., Tiner 1987; Fraye, Peters, and Pywell 1989; Fraye and Hefner 1991).

Status of Wetlands

Distribution

Numerous studies have described wetland distribution and abundance in the United States (see Hofstetter 1983; Mitsch and Gosselink 1986); however, direct comparison of estimates is difficult because studies often used different wetland definitions and survey techniques and covered different time periods. Furthermore, wetland types were not equally represented because of the narrow focus of many inventories. For example, the Office of Technology Assessment (OTA) (1984:26) used data derived from Shaw and Fredine (1956) to map the distribution of wetlands in the conterminous United States (Figure 3). However, Shaw and Fredine's (1956) data emphasized wetlands that were important to waterfowl. Consequently, maps based on Shaw and Fredine's data may overlook small but important wetland types such as northern peatlands and western riparian wetlands.

Abundance

Inland, freshwater wetlands accounted for 95 percent of the estimated 41.8 million ha of wetlands in the conterminous United States in the mid-1980s (Dahl and Johnson 1991). Of these, 52.9 percent were forested, 25.1 percent were emergent, 15.7 percent were scrub-shrub, and 6.3 percent were nonvegetated (e.g., open ponds and aquatic-bed areas) (Dahl and Johnson 1991). Deepwater habitats of the lacustrine and riverine systems accounted for an estimated 25.5 million ha.

Mitsch and Gosselink (1986:36-38) provided a State-by-State summary of wetland abundance estimates. In a more recent study, Dahl (1990) compared

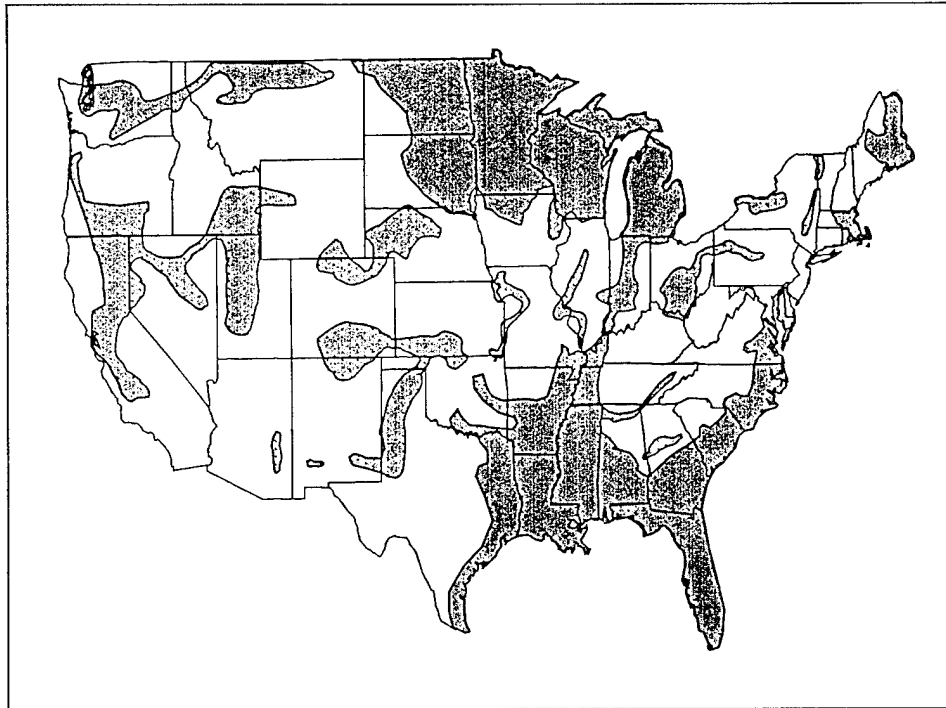


Figure 3. Distribution of wetlands in the conterminous United States (from Mitsch and Gosselink (1993:42), after Shaw and Fredine (1956)). Note: Coverage may be incomplete for small and/or isolated wetland types such as western riparian areas and small wetlands of the Southwest

the total surface area of wetlands and land in each State during the mid-1970s and the mid-1980s. States with the greatest percentage of their surface area as wetlands are Alaska (45.3 percent), Florida (29.5 percent), Louisiana (28.3 percent), Maine (24.5 percent), and South Carolina (23.4 percent); however, several other States (e.g., Minnesota, Texas, North Carolina, and Michigan) contain considerable wetland acreage (Dahl 1990:5). Wetland-acreage information is also available from the USFWS's NWI Project, but reports are limited to geographical areas that have been digitized (USFWS NWI Newsletter 6/92).

Status and trends

Estimates of wetland loss are not always comparable because wetland-inventory studies often had different objectives and used different classification schemes and inventory techniques. Nevertheless, most studies have indicated a rapid rate of wetland loss in the United States, at least prior to the mid-1970s. For example, the OTA (1984) estimated that 30 to 50 percent of the wetlands in the conterminous United States were lost from presettlement times to the 1970s. Mitsch and Gosselink (1986:41) estimated that 14.1 percent of inland freshwater marshes and northern peatlands and 9.6 percent of the southern deepwater swamps and riparian wetlands have been lost since the 1950s. In

the most recent study, Dahl and Johnson (1991) reported a net loss of 1.1 million ha of wetlands from the mid-1970s to the mid-1980s. Freshwater wetlands accounted for 98 percent of this loss, with most (54 percent) losses resulting from conversion of wetlands to agricultural uses (Dahl and Johnson 1991). Riparian wetlands are particularly vulnerable to these alterations, especially in the Southeast. For example, Wharton et al. (1982) reported that bottomland hardwoods were reduced by 60 percent because of agricultural conversion.

Wetland loss has not occurred evenly across the United States. Twenty-two States have lost ≥ 50 percent of their original wetlands since the 1780s, and 10 States—Arkansas, California, Connecticut, Illinois, Indiana, Iowa, Kentucky, Maryland, Missouri, and Ohio—have lost ≥ 70 percent of their wetlands (Dahl and Johnson 1991). From a regional perspective, the greatest rates of wetland loss occurred in the Lower Mississippi Alluvial Plain, the Pacific Mountains, the Gulf-Atlantic rolling plain, and the Gulf coastal flats (OTA 1984:96) (Figure 4). In absolute acreage, the greatest losses of wetlands occurred in the Lower Mississippi Alluvial Plain, the Gulf-Atlantic rolling plain, and the Upper Midwest (OTA 1984; Mitsch and Gosselink 1986:41).

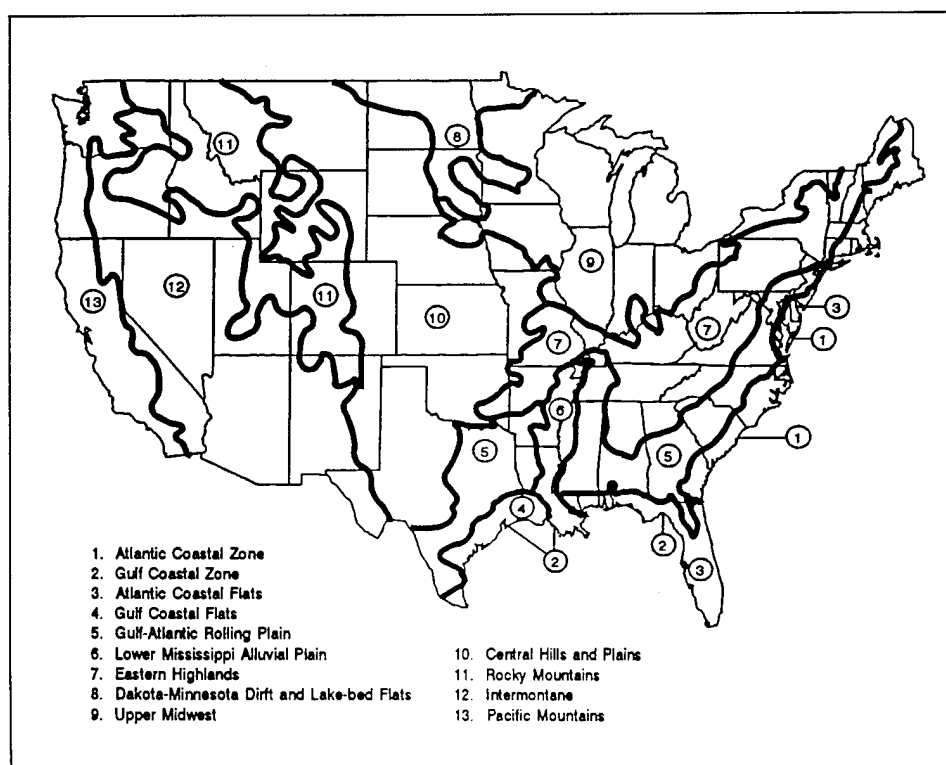


Figure 4. Physiographic regions used for regional analysis of national wetland-trends data (from the Office of Technology Assessment (1984:95)); boundary delineations were based on Land Surface Forms (Hammond 1964) and State boundaries

Influences and Alterations

Human influences

Human activities can significantly alter the ecology of wetland ecosystems. Wetland alterations result from land clearing and drainage (Dahl and Johnson 1991), hydrologic modifications such as stream channelization and dams (Mitsch and Gosselink 1986:124; Schneider, Martin, and Sharitz 1989), and various sources of pollution (White et al. 1991; Benson et al. 1991). Wetlands in agriculture-dominated landscapes may also be influenced by herbicide and pesticide runoff (Grue et al. 1986; Grue, Tome, and Swanson 1988; Sheehan et al. 1987; Tome, Grue, and Deweese 1991), nitrogen (N) and phosphorous (P) runoff (van der Valk et al. 1979; Neely and Baker 1989), cattle or sheep grazing (Behnke and Raleigh 1978; Platts and Raleigh 1984; Elmore and Beschta 1987; Clary and Medin 1990), burning practices (Hochbaum, Kummen, and Caswell 1985), and siltation. Some of these activities may be beneficial when conducted in a carefully planned and controlled manner (e.g., controlled grazing and prescribed burning); however, most agricultural operations are not designed for the benefit of wetlands.

Human activities in urban locations can also influence wetland ecosystems. For example, wetlands in or near urban areas are often influenced by municipal/industrial wastes and stormwater runoff. These sources of pollution frequently contain toxic materials, oils, trace-organic compounds, metals, and sewage effluent. Their effect on wetland biogeochemistry is poorly understood; nevertheless, created and natural wetlands are often used for the disposal and treatment of such material (Kadlec 1979a; Tilton and Kadlec 1979; Richardson and Schwegler 1986; Brown and Stark 1989; Hammer 1989; Olson 1992). Wetlands have been shown to be natural sinks for certain chemicals, particularly nutrients (Mitsch and Gosselink 1986:432). However, Richardson (1985) cautioned that after receiving wastewater for several years, a wetland may reach a level of saturation because of sediment adsorption and increased biomass. The reduced ability to retain pollutants is a process known as aging.

Regional influences

Wetlands are complex biological systems that exhibit regional and site-specific variability in geomorphology, water quality, species composition and richness, and biomass (Weller 1987; Hughes and Larsen 1988; Hughes et al. 1990; Omernik and Griffith 1991). Because of this variability, making broad generalizations about wetland ecology, management, and biodiversity is difficult, especially on a national scale. One solution is to group wetlands into ecological regions that have similar natural and anthropogenic influences. Scientists have developed several ecological-regionalization schemes (see Gallant et al. 1989); however, only a few schemes were designed specifically for aquatic ecosystems. Omernik (1987) and Omernik and Gallant (1990) described one such regionalization scheme. Some important characteristics (natural and human-related) of each region are summarized in Table 4.

| Table 4 Natural and Human-Related Characteristics of Omernik's Aggregations of Ecoregions of the Conterminous United States ¹ | | | |
|--|--|---|---|
| Section | Natural and Human-Related Characteristics ² | Aquatic Habitats | Environmental Stressors |
| Northern Predominately Glaciated Region | | | |
| Nonagricultural | Predominately forested; land-surface form mainly plains in Upper Midwest and hills and mountains in the Northeast; high-quality surface waters (i.e., low turbidity, dissolved salts, and nutrient concentrations); many lakes have low alkalinity levels; watersheds poorly defined in many areas | Continental glacial lakes; perennial effluent streams; wetlands mainly bogs, forested and scrub-shrub swamps, and inland freshwater marshes | Primarily related to silviculture and recreation activities, and locally by mining, agriculture (e.g., cranberry bogs), urbanization, and industrialization |
| Mixed land use | Generally warmer with longer growing season than nonagricultural section; surface water and wetland-quality potentials highly variable; watersheds poorly defined in many areas | Numerous wetlands of same types as in nonagricultural section | Predominately related to agricultural activities (cropland, grazing, and dairy cattle concentrations), and locally by atmospheric deposition, urbanization, and industrialization |
| Agricultural | Humid to subhumid climate with cool/cold winters and hot summers; mostly flat to irregular plains; very high percent of area in nonirrigated cropland; relatively poor-quality surface waters (e.g., high nutrient concentrations, suspended sediments, and dissolved solids); watersheds poorly defined in many areas | Mostly glacial lakes; wetlands sparse in heart of "Corn Belt" but numerous (mostly inland freshwater marshes) elsewhere; low gradient, effluent streams | Primarily associated with agricultural activity (including drainage, pesticide and herbicide applications, livestock operations, and soil erosion), and locally by urbanization and industrialization |
| Central and Eastern Predominately Forested Hills and Mountain Region | | | |
| No section divisions | >50-percent forest cover in most areas; forest type is mainly oak/hickory but some areas dominated by pine; surface-water quality is highly variable because of differences in soils and geologic characteristics | Most wetlands associated with rivers and reservoirs (i.e., riparian wetlands); streams mainly effluent with moderate to high gradients | Agriculture, silviculture, urbanization, hydrologic modifications (e.g., dams and stream channelization), mining (mostly coal), and atmospheric deposition |
| Sheet 1 of 4 | | | |
| ¹ Source: after Omernik (1987), Omernik and Gallant (1990), and unpublished mimeograph (J. M. Omernik, 10 May 1989). ² Listing is intended to give an understanding for why each region is distinguished from others; however, the listing is not intended to be exhaustive nor to provide a consistent list of factors for comparison among regions or sections (i.e., not all characteristics are important in every region). | | | |

Table 4 (Continued)

| Section | Natural and Human-Related Characteristics | | Aquatic Habitats | Environmental Stressors |
|---|--|--|---|---|
| South Central and Southern Humid, Mixed Land-Use Region | | | | |
| No section divisions | Extensive areas of forests (mostly pine but some mixed forests); hardwoods predominate in lowland areas; hot humid climate with long growing seasons; watersheds generally well defined, except in areas of karst topography; stream and reservoir problems common (e.g., high turbidity and low dissolved oxygen) | | Numerous perennial streams; more wetland acreage than in Central and Eastern Forested Hills and Mountain Region | Agriculture and silviculture activities, and locally by urbanization and industrialization |
| Subhumid Agricultural Plains Region | | | | |
| Northern | Mosaic of nonirrigated cropland and grazing land; cold winters and hot summers; usually <150 frost-free days; generally high concentrations of nutrients, alkalinity, suspended sediment, and dissolved solids are common in lakes and streams | | Most lakes and wetlands in shallow "prairie potholes" (glacial formed in MT, ND, and SD; wind-formed in the Sandhills of NE); most streams intermittent/ephemeral | Related to agriculture, including grazing |
| Southern | Land use predominately nonirrigated cropland and high-quality grazing land; mild to cool winters and very hot summers; usually >150 frost-free days; generally high concentrations of nutrients, alkalinity, suspended sediment, and dissolved solids in lakes and streams | | Most lakes are man-made reservoirs; few wetlands except along major rivers; most streams are intermittent or ephemeral | Related to agriculture, including grazing |
| Western Xeric Region | | | | |
| Semiarid | Most of section is in grazing or cropland; most water in streams and freshwater wetlands originates outside region or in outlier mountains and hills within the region; generally high-nutrient concentrations and suspended sediment in streams | | Few wetlands; wetland types predominately saline, associated with playas and wind-created depressions | Nonpoint source (NPS) impacts associated with agriculture activities, and locally by mining |

Sheet 2 of 4

Sheet 2 of 4

| Table 4 (Continued) | | | |
|---|--|--|--|
| Section | Natural and Human-Related Characteristics | Aquatic Habitats | Environmental Stressors |
| Western Xeric Region (Continued) | | | |
| Arid | Generally poor to no grazing potential, and very poor to no nonirrigated cropland potential; intensive irrigated agriculture locally near major water sources, particularly south of 240 frost-free-day isoline; most water in streams and freshwater wetlands originates outside region or in outlier mountains and hills within the region | Few wetlands; wetland types predominately saline, associated with playas and wind-created depressions | NPS stressors primarily associated with sparse grazing activity; locally by mining, oil and gas extraction, and irrigated agriculture; salinity is a major water quality problem |
| Western Forested Mountain Region | | | |
| No section divisions | Many areas of high-quality coniferous forests; generally high-quality water resources, particularly at higher elevations (e.g., high-gradient streams and alpine lakes); well-defined watersheds | Much variability in aquatic resources type, quality, and quantity related to elevation, exposure, latitude, and proximity to the coast | Primarily related to silviculture activities; locally by grazing, mining, recreation, and atmospheric deposition |
| Unique Alluvial and Coastal Plains Regions | | | |
| Central California Valley | Hot, dry climate; arid to semiarid potential natural vegetation types; perennial streams are influent and originate in mountains outside region; much of flat terrain is in irrigated agriculture | Many to most natural wetlands have been drained for agriculture or urban development | Primarily related to agriculture activities (mainly drainage, irrigation/salinity, chemical application, and cultivation/erosion) |
| Willamette Valley | Wet winters, dry summers | Most natural wetlands have been drained for agriculture (predominately nonirrigated) | Mostly related to agriculture activities, and locally by industrialization and urbanization |
| Western Gulf Coast Plain | Very hot summers and mild winters; precipitation evenly distributed throughout year; potential natural vegetation in two-thirds of section is grassland | Wetland types mainly forested and scrub-shrub swamps, and tidal freshwater marshes | Mostly related to agriculture activities, oil and gas exploration and production, and locally by industrialization and urbanization |
| Sheet 3 of 4 | | | |

Table 4 (Concluded)

| Section | Natural and Human-Related Characteristics | Aquatic Habitats | Environmental Stressors |
|---|--|--|---|
| Unique Alluvial and Coastal Plains Regions (Continued) | | | |
| Mississippi Alluvial Plain | Potential natural vegetation is southern floodplain forest; about one-half of section has been drained for cropland, remainder is in bottomland hardwoods | Wetlands mainly forested and scrub-shrub swamps, and tidal freshwater marshes | Industrial and municipal waste discharges and agriculture practices; also extensive channelization and flood control structures (e.g., levees and floodwalls) |
| Florida Coastal Plain | Potential natural vegetation mostly southern mixed forest; most of section in forest, but extensive areas have been cleared and drained for agriculture and urban/industrial development; watersheds are very poorly defined in many areas because of karst topography or flatness of terrain; frost-free period is 270-365 days | Numerous wetlands including bogs, coastal salt and inland freshwater marshes, and mangrove, scrub-shrub, and forested swamps | Agriculture (primarily for raising subtropical fruits and winter vegetables, and grazing); locally by urbanization and industrialization |
| Middle Atlantic Coastal Plain | Land use predominately forest with some cropland and pasture; frost-free period is 180-270 days | Wetland types mainly forested and scrub-shrub swamps, and coastal salt marshes | Silviculture and agriculture; locally by urbanization |
| <i>Sheet 4 of 4</i> | | | |

Wetland alterations

Keddy (1983) reported that wetland ecosystems are influenced by three main factors: water level, nutrient status, and natural disturbances. Human-induced impacts can modify these factors and lead to wetland alterations (Figure 5). These alterations can be grouped into three types: biological, chemical, and physical (Table 5). Biological alterations frequently result from management that maximizes specific wetland values (e.g., harvesting or removal of natural biota), although introduction or invasion of nonnative species (e.g., carp and purple loosestrife) can also cause biological alterations (NRC 1992a:277). Chemical alterations occur through point and nonpoint sources of pollution such as agricultural runoff (e.g., nutrients, pesticides, and herbicides), wastewater from sewage-treatment systems and mining operations, irrigation-caused contaminants (e.g., selenium and boron), and oil-related discharges and spills. Water quality problems resulting from such pollutants are becoming more widespread, especially in the western United States (Ratti and Kadlec 1992).

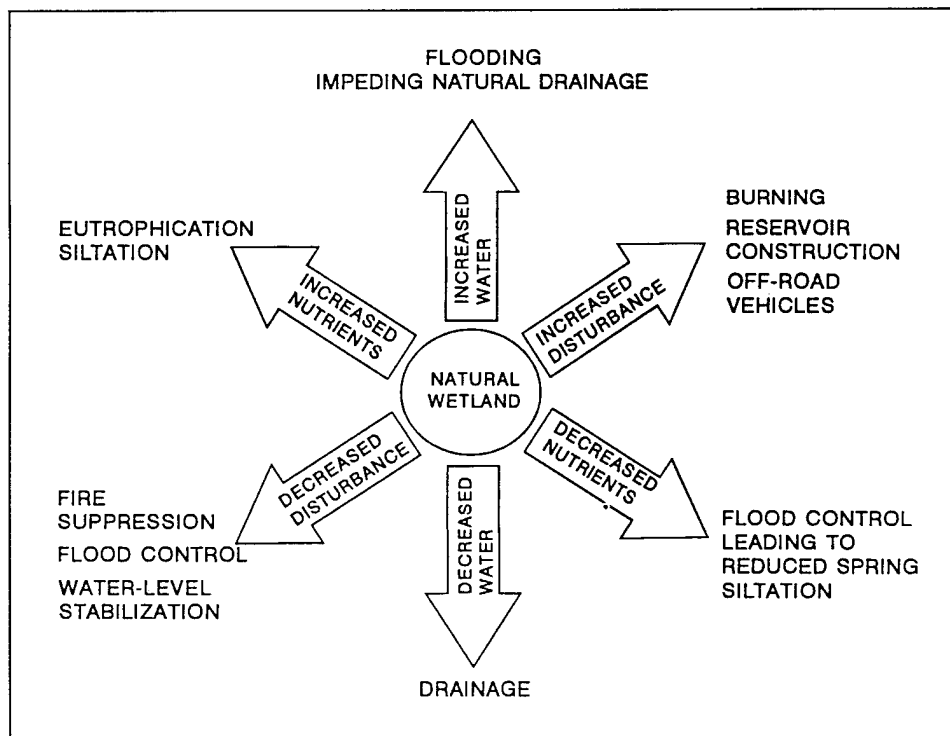


Figure 5. Model of human-induced impacts on wetlands, including effects on water level, nutrient status, and natural disturbance. By either increasing or decreasing any of these factors, wetlands can be altered (from Keddy (1983); copyright 1983 by Springer-Verlag, reprinted with permission)

Table 5
Types of Alterations to Wetlands¹

| Category | Description or Example of Alteration |
|---|---|
| Biological Alterations | |
| Grazing | Consumption and compaction of vegetation by either domestic or wild animals |
| Disrupting natural populations | Harvesting or removal of natural vegetation or animals; introduction of nonnative plants and animals |
| Competition | Competition for food and/or space, especially during the reproductive period, could result in changes in species composition because of the dominant role of certain species |
| Disease | Diseases, especially plant pathogens, can alter the composition of wetlands for extended periods |
| Chemical Alterations | |
| Changing nutrient levels | Increasing or decreasing levels of nutrients within the local water or soil system; forcing changes in the wetland plant community |
| Introducing toxics | Adding toxic compounds to a wetland either intentionally (e.g., herbicide treatment to reduce vegetation) or unintentionally, adversely affecting wetland plants and animals |
| Physical Alterations | |
| Draining or filling | Removing the water from a wetland by ditching, tilling, pumping, etc., or adding material to change the bottom level of a wetland or to replace the wetland with dry land |
| Excavating | Dredging and removing soil and vegetation from a wetland |
| Diverting water away | Preventing the flow of water into a wetland by removing water upstream, lowering lake levels, or lowering groundwater tables |
| Clearing | Removing vegetation by burning, digging, application of herbicides, scraping, mowing, or otherwise cutting |
| Flooding | Raising water levels, either behind dams or by pumping or otherwise channeling water into a wetland |
| Diverting or withholding sediment | Trapping sediment through construction of dams, channelization, or other types of projects, thereby inhibiting the regeneration of wetlands in natural areas of deposition such as deltas |
| Shading | Placing pile-supported platforms or bridges over wetlands, causing vegetation to die |
| Conducting activities in adjacent areas | Disrupting the interactions between wetlands and adjacent land areas, or incidentally impacting wetlands through activities at adjoining sites |
| Trampling and compaction | Onsite trampling of wetland vegetation and compaction of wetlands by foot traffic and off-road vehicles |
| ¹ Source: after The Conservation Foundation (1988:15) and National Research Council (1992a:278). | |

Physical alterations are often the most destructive because they frequently eliminate or significantly modify topography and hydrology (NRC 1992a:277). Common physical alterations in wetlands include (a) draining, dredging, and filling; (b) modification of hydrogeomorphology; and (c) mining and mineral extraction. However, the most significant historical loss of wetlands has resulted from wetland drainage and conversion to other uses, especially agriculture (Tiner 1984; Dahl 1990). These activities were most prevalent in the fertile soils of the prairie-pothole region and in the panhandle area of Texas, although the most rapid changes in the last two decades have occurred in the bottomland-hardwood forests of the Mississippi River alluvial floodplain (Mitsch and Gosselink 1986:419). Urbanization and industrial development also contribute to significant wetland losses through draining and filling.

Biological, chemical, and physical alterations often occur together, and their collective impact may well be synergistic (NRC 1992a:227). Furthermore, impacts may be cumulative in space or time. For example, the cumulative impact of local and regional perturbations can result in reduced potential for wetland restoration and may threaten the integrity of entire landscapes (NRC 1992a:278) and associated wildlife (Harris 1988; Weller 1988). Moreover, impact evaluations usually focus on proposed activities at individual sites and often fail to consider cumulative impacts at the landscape level (Risser 1988; Gosselink and Lee 1989; Gosselink et al. 1990a; Gosselink, Lee, and Muir 1990; NRC 1992a:279).

Impacts can also be described according to their timing, duration, and extent. Direct impacts are caused by specific activities and occur at the same time as the activities. Indirect or secondary impacts are also caused by specific activities, but their effect is later in time or farther removed. Permanent or temporary impacts indicate whether a wetland restores itself naturally after suffering perturbations; whereas, short- or long-term impacts indicate the length of time an impact takes to reveal itself after the activity occurs. A single activity may have temporary and permanent impacts, as well as short- and long-term impacts, simultaneously (OTA 1984).

5 Wetland Ecology

The probability of success in managing freshwater wetlands for the benefit of biodiversity can be increased by understanding basic principles of wetland ecology. For example, most freshwater marshes are dynamic systems that exhibit annual and seasonal changes in water levels and vegetative characteristics. These fluctuations are essential to nutrient cycling, decomposition, and maintenance of long-term productivity. In most freshwater marshes, these fluctuations result in highly productive systems where spatial heterogeneity is high and life cycles and food chains are complex. These characteristics subsequently influence the biodiversity associated with a freshwater marsh.

The following review synthesizes basic principles of freshwater-wetland ecology as they apply to understanding and managing interior wetlands of the United States. Although emphasis is on interior wetlands, many of the concepts and principles also apply to freshwater wetlands located in coastal areas. Moreover, brackish and even saline wetlands may show similar structural and biological patterns (Weller 1987:4). For more detailed information on wetland ecology, the reader should consult Good, Whigham, and Simpson (1978), Niering (1985), Weller (1987), Chabreck (1988), and Mitsch and Gosselink (1986, 1993).

Geomorphology and Hydrology

Most wetland basins were created by dynamic physical forces such as tectonic action, water and ice movement, soil movement and deposition, freezing and thawing action, and even meteorites (Weller 1987:7; Hammer 1992:41-62). For example, glacial action formed the Prairie Pothole region of the North Central United States and Canada (Winter 1989); tectonic action and changes in water flow resulted in wetland complexes of the Intermountain West (Ratti and Kadlec 1992); deposition of alluvial material formed river deltas like the McKenzie and the Mississippi Delta; wave and ice action created and maintained the once vast marshes located along the Great Lakes and Manitoba's giant-prairie lakes (e.g., Lake Manitoba and Lake Winnipeg); the dynamic nature of rivers (i.e., meandering, flooding, etc.) resulted in oxbow lakes/wetlands and riparian areas (Weller 1987:9-10); and continuous freezing and thawing were responsible for tundra wetlands and some alpine wetlands. The origin of some wetlands remains unclear, e.g., playa wetlands of the United States southern high plains (Haukoos and Smith 1992).

Although physical forces and geological conditions determine the occurrence of wetland types, their ultimate nature and morphology are influenced by the interaction of biotic components and hydrology. Plants are probably the most important biological influence because they control productivity, provide substrate, slow water movement, stabilize soils, create microclimates, and influence soil formation (Weller 1987:11). However, other organisms are also important. For example, invertebrates (especially detritivores and shredders) further influence soil-processing functions, and mammals such as beaver,¹ muskrat, and nutria can affect water levels, movement of soil, and vegetation. Nevertheless, hydrology and water chemistry (e.g., pH, alkalinity, and salinity) are probably the most important factors controlling structure and function of wetland ecosystems (Duever 1990).

Water source and hydrodynamics modify and determine the chemical and physical properties of the substrate, which subsequently influence biotic components of the wetland or riparian area (Gosselink and Turner 1978). However, ecosystem processes (e.g., decomposition, nutrient cycling, and productivity) also influence properties of the substrate and, in some cases, can modify water chemistry and hydrodynamics. Hydrology is further influenced by energy level of the ecosystem, which affects the export of toxins, nutrients, sediments, and organic matter. Wicker et al. (1982:84) presented a conceptual model of this complex relationship (Figure 6).

Hydroperiod, the seasonal pattern of water abundance in a wetland, varies regionally, locally, and by wetland type. The hydroperiod of a wetland is determined by its water budget, soil contours, and subsurface conditions. Major inputs into the water budget include precipitation, surface runoff, channelized flow (e.g., rivers and streams), groundwater, and tides in coastal areas. Major outputs in the water budget include evapotranspiration, surface outflows, groundwater outflows, and tides. Because hydrologic inputs are often responsible for the main transport of nutrients into wetlands, hydrology can have a major influence on productivity. However, the effect of water levels on net primary productivity for all wetland types is mostly unknown (Richardson 1979:141). Nevertheless, basic hydrologic principles apply in most cases (Mitsch and Gosselink 1986:79):

- a. Hydrology leads to a unique vegetation composition but can limit or enhance species richness.
- b. Primary productivity in wetlands is enhanced by flowing conditions and a pulsing hydroperiod and is often depressed by stagnant conditions.
- c. Organic accumulation in wetlands is controlled by hydrology through its influence on primary productivity, decomposition, and export of particulate matter.

¹ See Appendix B for scientific names of animals named in text.

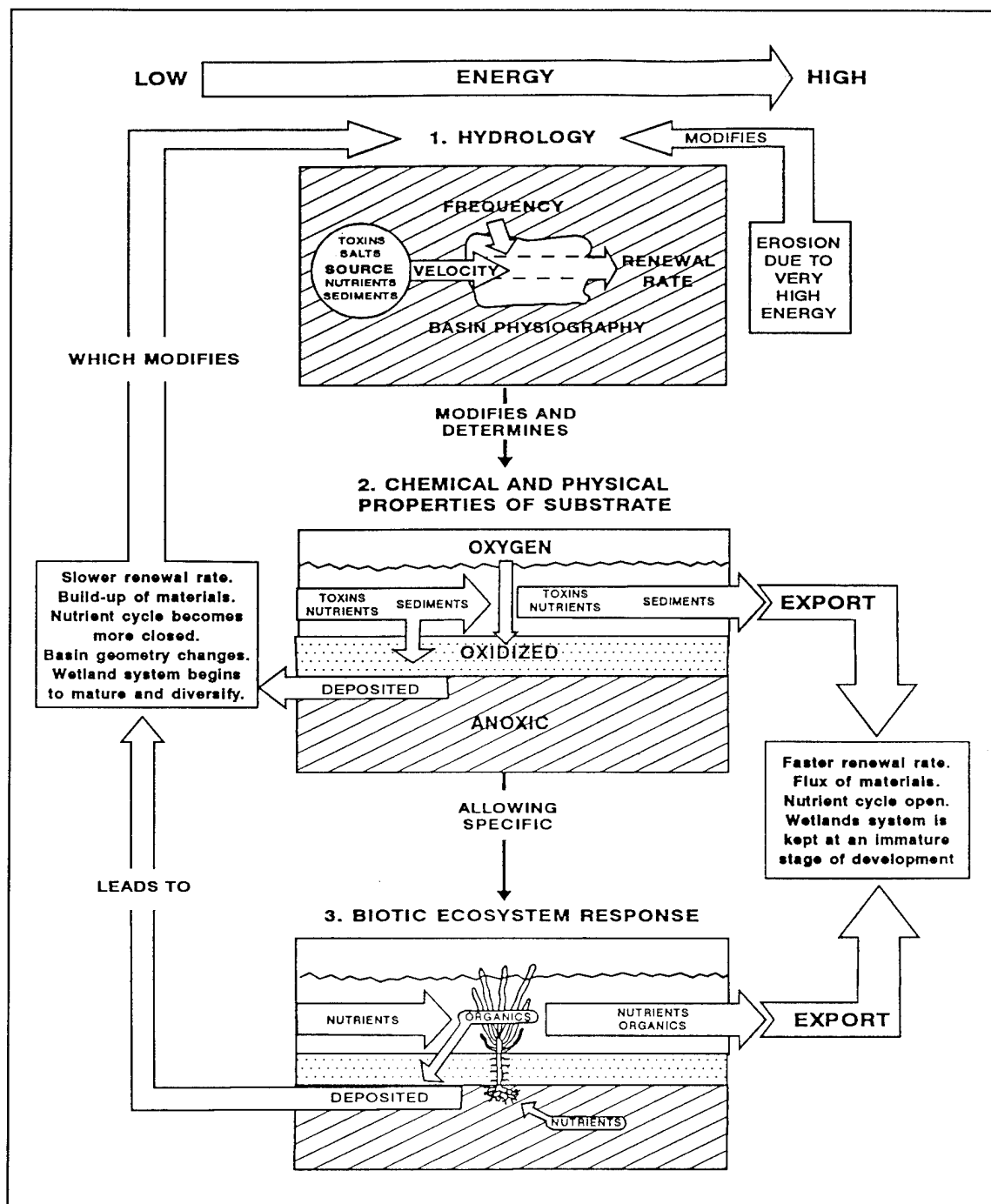


Figure 6. Conceptual model of hydrology and its relationship to other components in a wetland ecosystem (from Wicker et al. (1982:84); reprinted with permission of Coastal Environments, Inc.)

- d. Nutrient cycling and nutrient availability are both significantly influenced by the hydrologic conditions.

Although hydrology is often considered the driving force behind wetland ecology, it is a dynamic parameter that can be difficult to quantify. Thus, secondary characteristics such as soil and vegetation are often used to describe and define wetlands (Golet 1991).

Mitsch and Gosselink (1986:55-87) provided a thorough review of hydrology and its role in wetland ecology. Other general reviews were presented by Weller (1987:11-13) and Duever (1990). Winter (1989) reviewed hydrologic studies of wetlands in the prairie region. Fredrickson and Batema (1992:9-10) described types of flooding and the importance of hydroperiod in lowland-hardwood wetlands. Lugo, Brison, and Brown (1990) provided reviews of hydroperiod and the influence of water on forested wetlands worldwide, both for specific systems (e.g., riverine forests and fringe wetlands) and forested wetlands in general. Mitsch and Gosselink (1986) also discussed the role of hydrology in specific types of wetlands: (a) tidal saltmarshes, (b) freshwater marshes, (c) northern peatlands and bogs, and (d) riparian wetlands. Leitch (1981) and Hubbard (1981) provided annotated bibliographies of wetland-hydrologic studies. Hook et al. (1988a) provided reviews of hydrologic and water quality values of wetlands, and Hook et al. (1988b) provided reviews of hydrologic impacts of management activities.

Wetland Soils

Wetland soils act as both a medium for chemical transformations and for primary storage of available nutrients for most wetland plants (Mitsch and Gosselink 1986:89). Soils also influence the pioneering rate of plants, plant survival, and the stability and durability of the substrate (Weller 1987:18). Furthermore, soils influence the composition of the plant community because of differences in drying rate and moisture-holding capacity.

Wetland soils generally can be classified as (a) mineral soil or (b) organic or peat soil (also called histosols). Brinkman and Van Diepen (1990) discussed the nature and worldwide distribution of mineral soils associated with wetlands. Mineral soils generally have <20- to 35-percent organic matter on a dry-weight basis (Mitsch and Gosselink 1986:89); however, technical definitions also include criteria based on conditions of saturation and percent-clay content (e.g., U.S. Soil Conservation Service 1975:13-14, 65; Cowardin et al. 1979:42-43). Mineral and organic soils also differ in several important physicochemical features (Table 6).

Table 6
Comparison of Mineral and Organic Soils in Wetlands¹

| Physicochemical Features | Mineral Soil | Organix Soil |
|--|---------------------------------|---------------------------------|
| Organic content, percent | Less than 20 to 35 | Greater than 20 to 35 |
| Organic carbon, percent | Less than 12 to 20 | Greater than 12 to 20 |
| pH | Usually circumneutral | Acid |
| Bulk density | High | Low |
| Porosity | Low (45-55 percent) | High (80 percent) |
| Hydraulic conductivity | High (except for clays) | Variable; tends to be low |
| Water-holding capacity | Low | High |
| Nutrient availability | Generally high | Often low |
| Cation-exchange capacity | Low, dominated by major cations | High, dominated by hydrogen ion |
| Typical wetland | Riparian forest, some marshes | Northern peatland |
| ¹ Source: Mitsch and Gosselink (1993:117); copyright 1993 by Van Nostrand Reinhold, reprinted with permission | | |

Biogeochemistry

Regardless of wetland-soil type, long-term saturation will usually cause anaerobic conditions, which subsequently affect transformation, transport, and storage of chemicals. While anaerobic conditions and the resulting biogeochemical processes are not unique to wetlands, the complex interrelationships among hydrology, biogeochemistry, and biotic responses cause certain processes to be more dominant in wetlands than in either terrestrial or deepwater ecosystems.

Saturated soil usually becomes progressively anaerobic with increasing depth; however, a thin layer of oxidized soil (sometimes only a few millimeters thick) is usually found near the surface at the soil-water interphase (Mitsch and Gosselink 1986:94). This layer plays a key role in the chemical transformations and nutrient cycling that occur in wetlands. For example, ammonium (NH_4^+) is oxidized in this layer through the process of nitrification (Figure 7). The resulting nitrate ion (NO_3^-) is not subject to immobilization by negatively charged soil particles (Atlas and Bartha 1981) and, consequently, can more easily be assimilated by plants (Kadlec 1979b; Mitsch and Gosselink 1986:95-98).

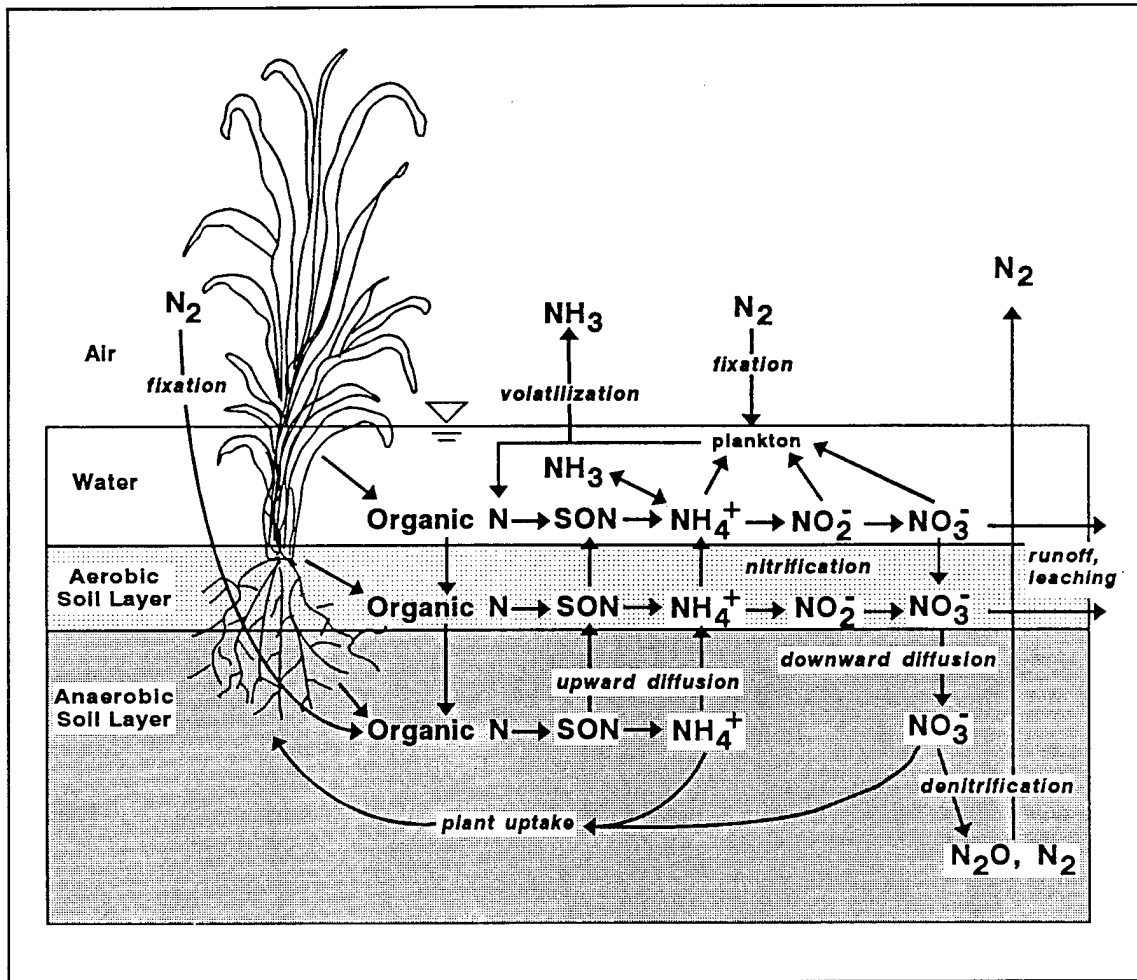


Figure 7. Nitrogen transformations in wetlands. SON indicates soluble nitrogen (from Mitsch and Gosselink (1993:128); copyright 1993 by Van Nostrand Reinhold, reprinted with permission)

Of the various chemical transformations that occur in wetlands, N and P transformations are often reported to be the most important because they potentially are limiting nutrients (Kadlec 1979b; Mitsch and Gosselink 1986:95-98, 105-106). However, other chemical transformations that occur within the anaerobic environment (e.g., iron, manganese, sulfur, and carbon) also affect the availability of minerals, and some, like hydrogen sulfide, can be very toxic to plants and microbes. Many chemical processes are mediated by microbial populations that are adapted to anaerobic conditions. Chemical transformations and biotic relationships are too complex to be described in detail here. Mitsch and Gosselink (1986:88-125), Richardson (1990), and Armentano and Verhoeven (1990) provided more detailed reviews. Other review and discussion papers on wetland biogeochemistry were presented in Hook et al. (1988a:253-351).

Biological Adaptations

Plants and animals that are regularly found in wetlands have evolved functional mechanisms to deal with the environmental stresses such as water depth, periodic drying and flooding, anoxia, and salinity. Microorganisms, most of which are relatively immobile, have developed some of the most interesting and important biological adaptations to these stresses. For example, many bacterial species are capable of switching from aerobic to anaerobic respiration (i.e., facultative anaerobes). However, some bacteria have become so specialized that they can grow only under anaerobic conditions (i.e., obligative anaerobes). These species rely on specific-electron acceptors other than oxygen (e.g., SO_2) to respire (Mitsch and Gosselink 1986:127-128). These adaptations are especially important because many microorganisms play a pivotal role in wetland biogeochemistry (i.e., they have significant roles in chemical transformations and ecosystem functions).

Protists in a saline environment also must deal with osmotic stress, i.e., water moving out of the cell and ions moving into the cell. The buildup of ions like Na^+ within the cell can denature important enzymes (proteins). Moreover, there is no evidence that protists can maintain water against the osmotic flow. Instead, protists in saline environments have developed a salt-adapted cell or maintain the osmotic balance by an active-transport system (i.e., Na-K pump). Some protists have specially adapted enzymes that function in higher saline conditions.

Wetland plants often have their upper portions in an aerobic environment and only their roots in an anoxic environment; consequently, their key adaptations involve structures that allow gas exchange down to the roots and stem. Most submergent and emergent species have air spaces (aerenchyma) in roots and stems that allow diffusion of oxygen from the aerial portion of the plant into the roots. Moreover, when hypoxia is moderate, oxygen diffusion through many wetland plants into the roots is apparently large enough to supply not only the roots, but also to diffuse out into the adjacent anoxic soil and create an oxidized rhizosphere (Mitsch and Gosselink 1993:173). This oxidized zone is important for the transformation and absorption of nutrients such as nitrogen (see Biogeochemistry). In contrast, very few woody species are successfully adapted to wetland conditions. The few species that have adapted (e.g., red mangroves, baldcypress, black tupelo, and willows) generally produce adventitious roots that take advantage of better oxygen conditions above the anoxic zone.

Both salinity and anoxic conditions are important stressors for wetland plants, but water depth is perhaps the dominant physical factor influencing the kind of adaptations required of plant species if they are to establish, survive, and reproduce on a wetland site. Various groups of plants have evolved different strategies for different water depths. Based on these strategies, wetland plants are classified into four groups (emergents, floating-leaf, submergent, and floating), although some researchers consider moist-soil species to be a unique,

| Table 7 Plant Life-Forms in Freshwater Wetlands of the United States¹ | | |
|---|--|---|
| Life-Form | Characteristics | Examples |
| Emergents | Roots and often bases of plants in wet soil or water part or all of their life; provide vital structure in wetlands; act as nutrient pumps | Cutgrass, sedges, whitetop, threesquare, cattail, bulrush |
| Floating-leaf | Rooted in deeper water; tend to send up broad, floating leaves to the surface where photosynthesis takes place; nutrients move between leaves and massive tubers via long, flexible, slender stems | Water lilies, watershield, spatterdock, some pondweeds |
| Submergent | Generally rooted but have stems and leaves mostly if not entirely underwater; seem to be efficient at gathering light, even in murky water; act as nutrient pumps | Sago pondweed, widgeon grass, coontails, water milfoils |
| Floating | Not rooted; usually remain on the surface of the water; flowering plants with dangling roots that derive nutrients from the water | Duckweeds, water hyacinth |
| Moist-soil | Native herbaceous vegetation managed for wildlife benefit; seeds adapted to germination in moist soil (seasonally flooded emergent wetlands); important food source for waterfowl and other wetland wildlife | Beggarticks, smartweeds, barnyard grass, foxtail grasses, chufa |
| ¹ Source: after Weller (1987:20-23,29) and Fredrickson and Taylor (1982). | | |

additional life form (Table 7). Moist-soil plants, an important food source for many wetland-wildlife species (Fredrickson and Taylor 1982; Reid et al. 1989), typically grow in the hydrologic-transition zone; however, moist-soil plants are a difficult group to define.

Most animals are mobile and can move out of an anoxic or highly saline zone, but this may result in exposure to other environmental stressors such as higher temperatures and potential desiccation. Although it is easier to think in terms of single-stress factors, most animals must respond to a complex of environmental conditions. Hence, their adaptations may represent a compromise that allows them to tolerate several environmental demands (Mitsch and Gosselink 1986:140). Some examples of adaptations to anoxic conditions include (a) evolution of specialized organs such as vascularized swimbladders, (b) development of high concentrations of respiratory pigments or pigments with unusually high affinities for oxygen (e.g., midge larvae or bloodworms), (c) reduced activity and metabolic demand (e.g., crabs in a low-oxygen environment such as a tidal marsh at low tide), and (d) conversion to anaerobic metabolism (e.g., glycolysis in crabs).

The most common response to temperature stress is mobility. For example, animals may burrow deeper, move underneath a shelter, or migrate in response to daily or seasonal changes in temperature. Salinity may also be a problem. Adaptations to saline conditions include the evolution of specialized cells,

glands, and organs, e.g., lachrymal glands at the corner of the eye or at the base of the avian bill; renal systems (e.g., kidneys) that incorporate filtering devices and countercurrent exchangers; and other structures such as rectal glands. In addition, estuarine/salt-marsh organisms must adapt to daily water-level fluctuations (i.e., tides) and hypersaline conditions because of evapotranspiration in shallow pools.

Biotic Components

A wetland community is composed of producers (autotrophs), consumers (holotrophs), and decomposers (saprotrophs). Primary producers are macrophytes and algae that transform solar energy into the potential form of fixed-carbon compounds (e.g., carbohydrates). Plants, with help from bacteria, also serve an important function by converting inorganic nutrients into organic forms (Murkin 1989). Furthermore, plants provide habitat that is important to the survival and reproduction of consumers within the system (Orth, Heck, and van Montfrans 1984; Weller 1987:25). Because of these functions, primary producers are often viewed as the critical link between consumers and resources of the wetland system (Murkin 1989).

Consumers can be divided according to trophic level (i.e., primary consumer, secondary, etc.), food habits (e.g., herbivores, carnivores, detritivores, and omnivores), and functional-feeding group (e.g., shredders, collectors, scrapers, predators, and parasites). Detritivores are perhaps the most important consumer group influencing the cycling of nutrients and flow of energy through a wetland system. For example, shredders (e.g., muskrats and some species of Plecoptera, Ephemeroptera, Trichoptera, Diptera, and Coleoptera) break down coarse-particulate-organic matter (CPOM) into smaller and smaller particles. Biological decay from oxidation of detritus by bacteria and fungi (decomposers) eventually results in fine-particulate-organic matter (FPOM), which is used by benthic-gathering and filtering collectors (e.g., clams, springtails, and some species of Ephemeroptera, Trichoptera, Diptera, and Coleoptera). These species subsequently support populations of predatory organisms and so forth through the food web. Relationships among these groups and other functional components can be complex, even if one restricts the discussion to invertebrates (Figure 8).

The study of food chains (or food webs) in wetland ecosystems is further complicated by spatial and temporal diversity of both producers and consumers (Clark 1978; Crow and Macdonald 1978; Murkin 1989). Nevertheless, certain functional groups are known to be critical to ecosystem processes such as decomposition and nutrient cycling (Cummins and Merritt 1984; Murkin and Batt 1987; Murkin and Wrubleski 1988; Murkin 1989), which subsequently influence species richness in wetland systems. In terms of biodiversity, most wetlands function best (i.e., high productivity and species richness) when all components are present and effective (Weller 1987:25).

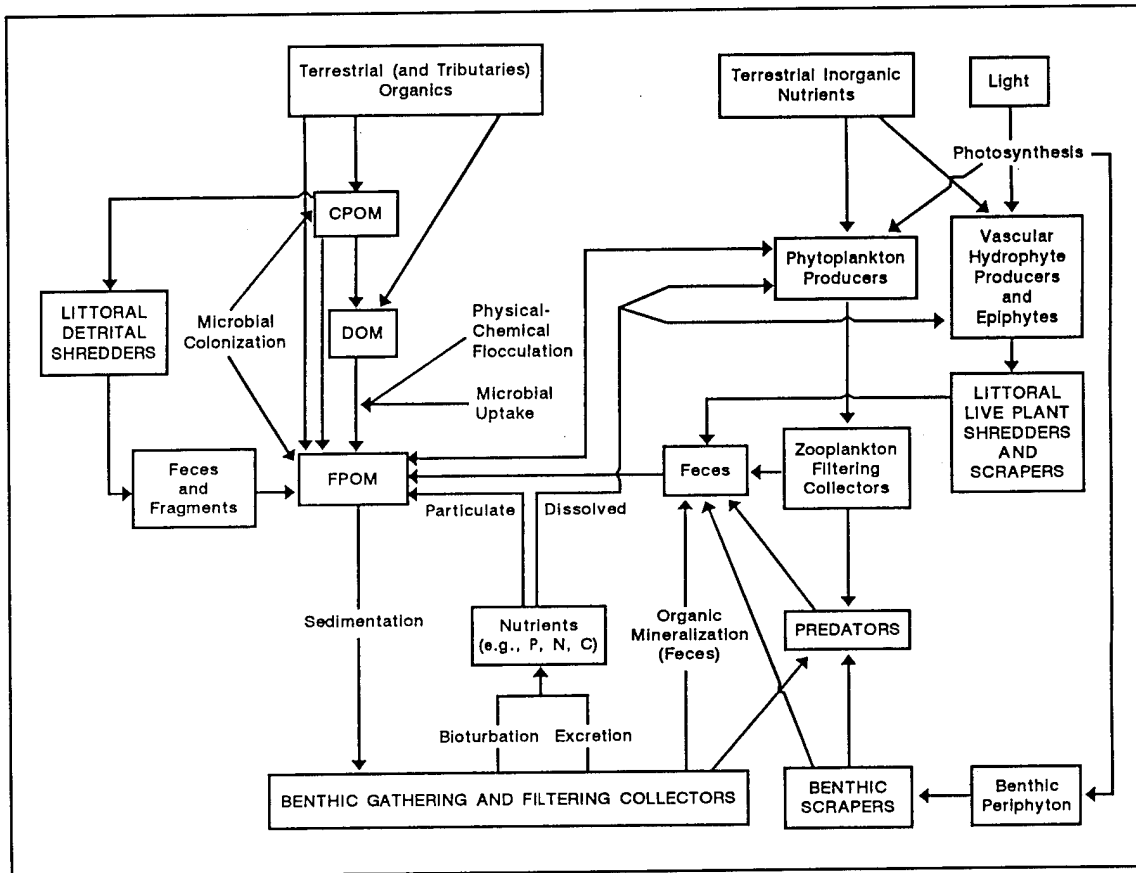


Figure 8. Nutritional-resource categories and invertebrate-functional-feeding-group categories in lentic ecosystems (from Cummins and Merrit (1984:63); copyright 1984 by Kendall/Hunt Publishing Company, reprinted with permission)

Nutrient Cycling

Nutrient cycling is influenced by the age of the system, temperature, seasonality of the growing season, hydrology, and other environmental factors (Mitsch and Gosselink 1986:159-165; Kadlec 1986a,b, 1987). In some cases, anthropogenic factors (e.g., agricultural or wastewater runoff) and biotic factors (e.g., nitrogenous waste from concentrations of birds and mammals) also are important (Weller 1987:28; Neely and Baker 1989). In general, wetlands that are "open" to hydrologic transport tend to be nutrient-rich (eutrophic) and have a loose, rapid nutrient cycle; whereas "closed" wetlands tend to be nutrient-poor (oligotrophic) and rely upon tight intrasystem cycling of nutrients, which often is extremely slow (Table 8). However, the relationship among hydrology, nutrient inputs, productivity, decomposition, export, and nutrient cycling can be complex (see Kadlec 1979b, 1986a,b; Mitsch and Gosselink 1986:82).

Table 8
Characteristics of High-Nutrient (Eutrophic) and Low-Nutrient (Oligotrophic) Wetlands¹

| Characteristics | Low-Nutrient Wetland | High-Nutrient Wetland |
|--|--|--|
| Inflows of nutrients | Mainly precipitation | Surface and groundwater |
| Nutrient cycling | Tight, closed cycles; adaptations such as carnivorous plants and nutrient translocations | Loose, open cycles; few adaptations to shortages |
| Wetland as source or sink of nutrients | Neither | Either |
| Exporter of detritus | No | Usually |
| Net primary productivity | Low (100-500 g/m ² /year) | High (1,000-4,000 g/m ² /year) |
| Examples | Ombrotrophic bogs; cypress dome | Floodplain wetland; many coastal marshes |

¹ Source: Mitsch and Gosselink (1993:161); copyright 1993 by Van Nostrand Reinhold, reprinted with permission.

The bulk of P and N, key nutrients for plant growth, apparently comes from and returns to the sediment through nutrient imports and intrasystem cycling (Kadlec 1979b). Most nutrients in wetlands are permanently tied up in sediments, peat, or plant biomass, and are lost from ecosystem cycling as peat deposits or organic exports (Mitsch and Gosselink 1986:123). Intrasystem cycling depends on emergent and submergent plants, which extract nutrients (especially P) from deeper anaerobic sediments and return them, by decomposition, to subsurface sediments and, to some degree, the water (Prentki, Gustafson, and Adams 1978; Weller 1987:29). Consequently, macrophytes are often referred to as nutrient pumps. Although vascular plants are important to intrasystem cycling, there is conflicting evidence regarding whether they are effective in regulating and reducing nutrients in surface waters (van der Valk et al. 1979; Kadlec 1979b, 1987). There is also disagreement about whether wetlands act as sources, sinks, or transformers of nutrients (Mitsch and Gosselink 1986:113, 119-120) and how this relates to seasonal patterns of nutrient uptake and release.

Most wetlands act as nutrient sinks during the growing season, i.e., there is increased microbiological immobilization of nutrients, and there is a high rate of uptake of nutrients by aquatic macrophytes, algae, and epiphytes. Prior to senescence, plants translocate nutrients back to the roots and rhizomes; however, a substantial portion of the nutrients may be lost to the water through litter fall and subsequent leaching. Hence, wetlands are potential nutrient sources in the fall and early spring (van der Valk et al. 1979). The actual amount of nutrient export (if any) and the timing of export depend on wetland characteristics such as vegetation type, hydrologic regime, temperature, humidity, etc.

Wetland managers should understand nutrient cycling and seasonal pulses so that they can control the potential loss of nutrients, especially in nutrient-poor systems. For example, planned drawdowns and reflooding are common practices in wetland management; however, improper timing and duration could potentially lead to a net export of nutrients and possible N deficiencies (Kadlec 1979b). Also of concern is excessive N and P inputs from agricultural runoff and other sources of pollution (Neely and Baker 1989). These inputs can lead to eutrophication (Schindler 1974), which promotes algal blooms and reduced-oxygen conditions (Weller 1987:30).

Wetland Dynamics

Wetlands are dynamic systems that may exhibit daily (in tidal areas), seasonal, and short-term (3- to 10-year) changes. Seasonal changes typically result from variation in temperature, precipitation, and photoperiod, and are more evident at extreme latitudes and high altitudes. These changes influence wetland productivity and, subsequently, seasonal variation in species richness. For example, Weller (1987:51-53) described southern marshes as having fewer breeding-bird species than northern and midlatitude marshes; however, southern marshes support a large, diverse group of migrating- and wintering-bird species.

Seasonal changes in resource abundance strongly influence strategies of adaptation, i.e., habitat selection, breeding biology, migratory chronology, and general distribution of species. Nonmigratory species also must respond to this pulse. For example, invertebrates in northern climates have developed different strategies for over-winter survival. Some species have a short life cycle (<1 year) in which the next generation overwinters as eggs. Other invertebrate species have a longer life cycle (>1 year) in which intermediate stages (i.e., instars, larvae, and pupae) overwinter in deeper, nonfreezing sediments. Nonmigrating vertebrates also must respond to seasonal changes in temperature and precipitation. For example, a common behavioral response to winter is burrowing or the use of microsites (e.g., muskrat houses and beaver lodges).

Many wetlands are also subject to short-term (3- to 10-year) vegetative changes. Dominant forces include water-level fluctuations, herbivore activity (e.g., muskrats, nutria, and beaver), ice action, and, possibly, fire and nutrient turnover (Weller 1987:74). Like seasonal changes, short-term changes often follow a rather predictable pattern (especially in freshwater marshes of the Midwest). For example, as an emergent wetland becomes dry, the bottom sediments become more aerobic and soil structure is altered by leaching, podzolization, and cracking of the soil. There may be significant die-offs of invertebrates and fish, which contribute additional nutrients to the system. Depending on the degree of dryness, compositional changes in the invertebrate community also may occur (Voigts 1976; Murkin and Kadlec 1986; Bataille 1991).

Moist-soil conditions and increased air exchange stimulate the germination of plants from exposed seed banks. Damp, aerobic conditions tend to increase the decomposition rate and reduce soil toxicity (i.e., toxic substances that

accumulated in the soil during anaerobic conditions are oxidized to different forms that allow plants to regenerate) (Cook and Powers 1958). Moist-soil plants (primarily annuals) are usually the first to regenerate, with the vegetative composition being determined by the seed bank and existing microclimate conditions (van der Valk and Davis 1978; Smith and Kadlec 1983; Pederson and van der Valk 1984).

Vegetative responses to reflooding depend on timing, rate of flooding, degree of inundation, water chemistry, herbivore activity, energy of the system (e.g., wave action), and many other influences (Kadlec 1962; Meeks 1969; Millar 1973; Weller 1987:56; Merendino et al. 1990; Merendino and Smith 1991). Gradually, moist-soil plants are outcompeted by more persistent emergent species (e.g., cattail and bulrush), and then later submergent species (e.g., sago pondweed and watermilfoil) begin to regenerate. Continuous flooding and activity of herbivores tend to move the marsh towards an open-water community (Weller 1987:57). Plant-species richness and diversity begins to decline and, eventually, one or two water-adapted species (e.g., cattail) dominate the community. These spatial and structural changes in the plant community affect nutrient availability, food chains, substrate development, and habitat structure, which subsequently influence abundance, diversity, and composition of the animal community (Weller and Spatcher 1965; Voigts 1976; Nelson and Kadlec 1984; Burger 1985; Murkin and Batt 1987; Murkin 1989; Neckles, Murkin, and Cooper 1990; Murkin, Kadlec, and Murkin 1991).

Species richness and diversity generally are highest in large, shallow marshes with a ratio of emergent vegetation to open water ranging from 1:1 to 1:2 (Weller and Fredrickson 1974; Weller 1982, 1988:65). However, managers must remember that (a) most wetlands are dynamic and the cover:water ratio will vary naturally over time; (b) in many cases, achieving and maintaining a 1:1 ratio without extensive habitat manipulations may be difficult, which could result in management actions that damage the existing wetland; and (c) long-term productivity and health of the wetland are often achieved by allowing the system to follow natural patterns of change.

In addition to the cover:water ratio, the spatial distribution and size of openings may be important management considerations in large marshes (Kaminski and Prince 1981, 1984; Murkin, Kaminski, and Titman 1982; Ball and Nudds 1989). Interactions with other ecosystems in the landscape also are important. For example, adjacent upland habitats are vital to some species (e.g., dabbling ducks) to complete their life cycle. Consequently, the best management approach may be to preserve a heterogeneity of wetland types in combination with adjacent-upland areas to create habitat diversity, which promotes high-species richness (Weller 1982). A similar response can be achieved by maintaining several small marshes in different "successional" stages (Weller 1987:73).

Succession, in the classic sense, is often interpreted as "the replacement of plant species in an orderly sequence of development" (Mitsch and Gosselink 1993:190). In other words, ecosystems, including wetlands, contain distinct

plant communities in which change is autogenic (i.e., change is brought about by the biota) and directional (i.e., proceeding toward a stable, climax community) (Clements 1916; Odum 1971). In contrast, several scientists argued that distribution of plant species is governed by its response to the environment (allogenic succession) and the chance occurrence of propagules at the site. This alternate hypothesis is known as the individualistic hypothesis (Gleason 1917) or continuum concept (Whittaker 1967; McIntosh 1980). In this view, ecosystem change also occurs, but it is not directed toward a particular climax community. More recently, van der Valk (1981, 1982) applied this hypothesis to succession in temperate, North American wetlands. He suggested that the pattern of succession in wetlands is based on site variation, randomness, and plant-life history, and that environmental factors compose an environmental sieve that acts upon unique life-history characteristics of each plant. As the environment changes (e.g., water-level changes), so does the sieve and thus the species present. This is a useful concept for understanding wetland change in relation to wetland dynamics (Mitsch and Gosselink 1993:202).

Although the traditional view of succession has limited usefulness when applied to wetland dynamics, it can be used to compare the structural and functional properties of wetlands with other types of ecosystems that typically pass through different successional stages toward "maturity." For example, wetlands often are highly productive systems with open-mineral cycles and some export of production, which are attributes of young ecosystems. However, these same wetlands frequently are detrital systems where spatial heterogeneity is high and life cycles and food chains are complex, which are indications of mature ecosystems. Hence, wetlands exhibit characteristics of both immature and mature systems. Furthermore, both autogenic and allogenic processes are important in the pathway of development and in the final characteristics of mature-wetland ecosystems (Mitsch and Gosselink 1986:162).

Wetland Functions and Their Value

Much of the literature on wetland functions and values has been written by scientists from a noneconomic perspective (Leitch and Shabman 1988). The term "value" has often been used in an ecological sense to refer to functional processes such as primary production and energy flow. However, value is an anthropocentric term that should not be confused with ecological function (Mitsch and Gosselink 1986:393). A "function" describes what a wetland does, irrespective of any beneficial worth assigned by man; whereas a "value" is a subjective interpretation of the relative worth of some wetland process or product, i.e., the market or recreational value or cost (Hammer 1992:69).

Wetland functions

Wetlands are legally protected because many of the functions they perform are valuable to society (Table 9). One widely valued function of wetlands is the maintenance of intraecosystem and interecosystem integrity (e.g., providing habitat for fish, birds, and other wildlife). For example, although wetlands

Table 9
Functions of Wetlands and Their Value¹

| Functions of Wetlands | Value of the Functions of Wetlands |
|--|--|
| Store and/or convey floodwater | Reduce flood-related damage |
| Buffer storm surges | Reduce flood-related damage |
| Recharge groundwater | Maintain groundwater aquifers |
| Discharge groundwater | Maintain base flow for aquatic species |
| Stabilize shorelines | Minimize erosion damage |
| Stabilize streambanks | Minimize erosion damage |
| Detain/remove/transform nutrients | Maintain water quality |
| Detain/remove sediments | Maintain water quality |
| Maintain intraecosystem/interecosystem integrity | Maintain plant and animal populations Preserve endangered species Maintain biodiversity Provide renewable food and fiber products |
| Setting for cultural activities | Provide recreational opportunities Provide education/research opportunities Provide aesthetic enjoyment Preserve archeological and historical sites |
| ¹ Source: Smith (1993). | |

occupy only about 5 percent of the land surface in the conterminous United States (Tiner 1984; Dahl 1990), over 900 species of wildlife require wetland habitats at some stage in their life cycle (Hammer 1992:71). This includes greater than one-third of the Federally listed endangered and threatened plants and animals (NRC 1992a:265; Williams and Dodd 1979; Niering 1988; Feierabend 1992; Hammer 1992:70). Many more species are facultative users of wetland habitats.

Wetlands exhibit great diversity in both structure and function. Moreover, the structure and function of individual wetlands can, in even a few years of time, change significantly. For example, a semipermanent prairie-pothole may operate as a discharge wetland (i.e., a wetland receiving groundwater input) during periods of average to above-average precipitation; however, the same wetland may act as a recharge wetland during periods of extended drought. This spatial and temporal variation makes it extremely difficult to assess the functions of wetlands because "as functional diversity increases, so must the complexity of the assessment method" (Smith 1993:7).

Several procedures have been proposed for assessing (a) the ability of wetlands to perform functions and (b) the potential adverse effects of individual projects on those functions (see reviews in USEPA 1984; World Wildlife Fund 1992; Smith 1993; Solomon and Sexton 1993). Two procedures that have

received national attention are the Wetland Evaluation Technique (Adamus et al. 1987) and the Habitat Evaluation Procedure (USFWS 1980). The Wetland Evaluation Technique (WET) is a rapid-assessment method that estimates the "probability" that a function is performed to an unspecified degree, or magnitude. Habitat Evaluation Procedures (HEP) is another decision-making tool with broad applications. The HEP procedure focuses exclusively on biological functions (i.e., support of fish and wildlife) and uses an index to rate habitat quality, which can then be used to compare management and development scenarios. Other methods have been developed and used at the regional scale; however, no single method satisfies all regulatory, administrative, and technical requirements of the 404 Regulatory Program (Smith 1993).

Value of wetland functions

Wetland functions are valued for a variety of reasons (Table 9); however, not all functions are universally recognized or equally valued. For example, privately owned wetlands may have limited value to the individual landowner; however, these same wetlands provide social and economic benefits that are valued by society as a whole (NRC 1992a). Part of the problem arises because many goods and services provided by wetlands are difficult to allocate through economic markets (Leitch and Shabman 1988).

Recently, wetland scientists have put considerable effort into describing ecological and cultural values of wetlands (Greeson, Clark, and Clark 1979; OTA 1984; Sather and Smith 1984; Shabman and Batie 1988; Niering 1986; Hook et al. 1988a; Sather, Smith, and Larson 1990). There also is a national *Wetlands Values Database* (USFWS, St. Petersburg, FL), which contains nearly 15,000 records for citations pertinent to wetland values and functions (Stuber 1986; Wilkinson et al. 1987).

Functions can generally be grouped according to three hierarchical levels—population, ecosystem, and global (Mitsch and Gosselink 1986:393-406; Odum 1978). The values resulting from population functions are the easiest to identify and measure, e.g., animals harvested for food or their pelts; the waterfowl-recreational-hunting industry; harvest of timber and other vegetation; and survival of endangered and threatened species. At the ecosystem level, wetland functions are valued for reducing flood-related damage, maintaining groundwater aquifers and water quality, and providing aesthetic enjoyment. On a much broader scale, wetlands are valuable because they are important in the global cycling of nitrogen, sulfur, methane, and carbon dioxide. For example, carbon-fixation rates in marshes are more than double the rates for forests. Furthermore, and perhaps more importantly, wetlands immobilize significant quantities of sulfur, a major constituent in acidic precipitation.

Valuation methods

There are several well-accepted approaches to valuation of wetland goods and services (Lonard et al. 1981; USEPA 1984; Mitsch and Gosselink 1986;

Wilkinson et al. 1987; Shabman and Batie 1988; Leitch and Shabman 1988; Luzar and Gan, In Preparation), but there is disagreement among economists, ecologists, and resource managers about the best method (Whigham and Brinson 1990). The choice is influenced by the circumstances and wetland attributes being evaluated.

Wetland benefits can generally be divided into market and nonmarket goods and services. Economic assessment of marketed goods and services (e.g., timber and commercial-fish harvest) is relatively straightforward; it is based on observed market prices and capitalized values of revenue streams (Leitch and Shabman 1988). Economic assessment of nonmarket goods (i.e., goods and services not traded in the market place) involves estimating the price that would have been observed if market trading existed. Several methods are available, including the land-price method (Freeman 1979), travel-cost method (Freeman 1979; Randall 1981), contingent-valuation technique or willingness-to-pay approach (Cummings, Brookshire, and Schulze 1986; Whitehead 1990), replacement costs (Wilkinson et al. 1987; Mitsch and Gosselink 1986:414), and unit-day-value method (U.S. Water Resources Council 1983; USFWS 1985). In addition, there are several methods based on noneconomic standards such as existence value (Randall 1981; Leitch and Shabman 1988; Smardon 1988), embodied energy (Costanza 1980), and other attributes (Siden and Worrell 1979; Smardon 1988).

Socioeconomic analysis provides humans with a means of resolving conflicts among different wetland uses. This usually involves two kinds of evaluations: (a) determination of the ecological value of the area in question (e.g., the quality of the site compared with similar sites, or its suitability to support wildlife), and (b) comparison of the ecological value of the habitat against the economic value of some proposed activity that would destroy or modify it (Mitsch and Gosselink 1986:406-407). However, economic analysis of such activities usually considers not only current values but the cost of delaying the benefits from a resource into the future, i.e., the concept of discount rate (Pearce 1976; Pearce and Turner 1990). For example, if the discount rate is very high, then it makes economic sense to exploit a resource into extinction, regardless of the impact on future generations. Conversely, if the discount rate is set to almost zero, then immediate exploitation of a resource is difficult to justify in economic terms because the long-term cost of delaying the activity is minimal. Resource economists use the concept of discount rate to balance the economic value of long- and short-term benefits; however, the concept does not appear to be consistent with the ideas of conservation and sustainability (Pearce and Turner 1990:212). Nevertheless, the concept is frequently applied in everyday life, including decisions regarding natural resources. For example, does it make economic sense to drain a swamp, harvest the timber, and then plant soybeans? The answer depends on the concept of discount rate.

Socioeconomic analysis has also been used to value ecological damage from perturbations such as oil spills (Desaigues 1990:280). Damage-assessment models have been developed for coastal/marine and Great Lakes environments. Despite advances in valuation techniques, several generic problems remain: (a) wetlands are multiple-value systems; (b) the most valuable products of

wetlands are public amenities that have no commercial value for the private-wetland owner; (c) a wetland's value is related to its interspersion in the landscape, not to its size; and (d) commercial values are finite, whereas wetlands provide values in perpetuity (Mitsch and Gosselink 1986:407-408).

6 Biodiversity

Biological diversity (commonly termed biodiversity) is the variety of life and its processes (Hughes and Noss 1992). The goal of biodiversity management is to provide the conditions and ecological processes necessary for sustaining life, in all of its complexity (Landres 1992). Despite the economic and ecological importance of biodiversity to humans, the rate of ecosystem alteration has increased dramatically in the last 150 years (O'Connell and Noss 1992). These alterations have resulted in the loss and degradation of habitats, biotic assemblages, and ecological processes (Wilson 1988; Myers 1988; Hughes and Noss 1992). Although attention has centered on terrestrial ecosystems, biodiversity of wetland and deepwater ecosystems has also been reduced (Cairns and Lackey 1992; Hughes and Noss 1992).

Wetland managers are being asked to place more emphasis on biodiversity conservation, while simultaneously maintaining other wetland functions and values (Laubhan and Fredrickson 1993). To effectively meet this challenge, resource personnel must develop a better understanding of wetland ecology, as well as the life-history requirements of species that rely on wetlands. Consideration must not only be given to charismatic species (e.g., game, threatened, and endangered species) but also other organisms associated with wetland ecosystems. Providing the required resources for a constantly increasing group of target species may be one of the biggest challenges facing wetland managers today (Laubhan and Fredrickson 1993).

The biology and habitat requirements of wetland-associated game and furbearer species are well documented. In contrast, little is known about the life history and habitat requirements of many nongame species (Fredrickson and Reid 1986). Moreover, the total number of species of smaller organisms (e.g., insects and bacteria) in wetland systems is not even known, much less the effect management activities have on these species. Consequently, intensive management for biodiversity is a difficult process. On the other hand, ecological principles that apply to populations and communities of game species also apply to most nongame species (Temple 1986). Thus, many nongame species may already benefit from wetland-management activities designed to help waterfowl and furbearing mammals (Fredrickson and Taylor 1982; Rakstad and Probst 1985; Fredrickson and Batema 1992; Helmers 1992; Wentz and Reid 1992).

Biodiversity conservation involves more than increasing the number of species occurring in individual wetlands. The focus of management must change from individual wetlands and featured species to a larger scale approach that

considers wetland complexes, adjacent uplands, habitat corridors, and ecological processes. This new challenge will require (a) a complex and comprehensive approach to wetland management (Fredrickson and Reid 1986; Laubhan and Fredrickson 1993; Reid 1993; Parcels and Dunstan 1993), (b) carefully planned monitoring and assessment studies to guide management strategies and test predictions about wetland ecology and conservation biology (Holling 1978; Romesburg 1981; Weller 1987:87; Noss 1990; Murphy and Noon 1991; Noss et al. 1992; Ratti and Garton 1994), and (c) skilled managers who understand the principles of ecosystem management and conservation biology and are able to integrate this knowledge with more traditional information on ecology, management, and biopolitics of freshwater wetlands.

Biodiversity: A Primer

Biodiversity has recently emerged as an important public-policy issue; however, scientists have been interested in protecting the diversity of life for many years. The scientific literature is replete with diversity-related discussions (Hutchinson 1959; MacArthur 1965; Hurlbert 1971; Whittaker 1972; Peet 1974; Pielou 1975), including descriptions of wetland/riparian communities (Krull and Boyer 1976; Nudds 1983; Monda and Ratti 1988; Douglas et al. 1992). Many of these descriptions focus on numbers and/or distribution of organisms. Consequently, biodiversity is often interpreted as simply the number of plant and animal species and abundance of each species in a given area (e.g., an individual wetland). This is an incomplete and overly simplistic view of biodiversity.

Biodiversity has multiple levels of organization (Noss 1983; OTA 1987; Harris 1988; Scott et al. 1993) and includes structural, functional, and compositional components (Figure 9). Ecological and evolutionary processes are also an important part of biodiversity (O'Connell and Noss 1992). To understand biodiversity and its implications for land management, one must be aware of these factors and how they interact to affect communities and individual species (O'Connell and Noss 1992; Landres 1992). Moreover, this complexity should be considered when developing strategies to inventory, monitor, and assess biodiversity (e.g., Table 10).

Genetic diversity

Genetic diversity refers to the amount of variation at the molecular level among individuals of the same species (OTA 1987). It is frequently measured in terms of heterozygosity or polymorphism, or in terms of variation at the DNA level (see Ralls and Ballou 1992; Hedrick and Miller 1992). Within a population, maintenance of genetic diversity depends on mutation pressure, natural selection, gene flow, and genetic drift, with mutation and gene flow providing the ultimate source of new genetic variation (Wilson and Bossert 1971; Hartl 1988).

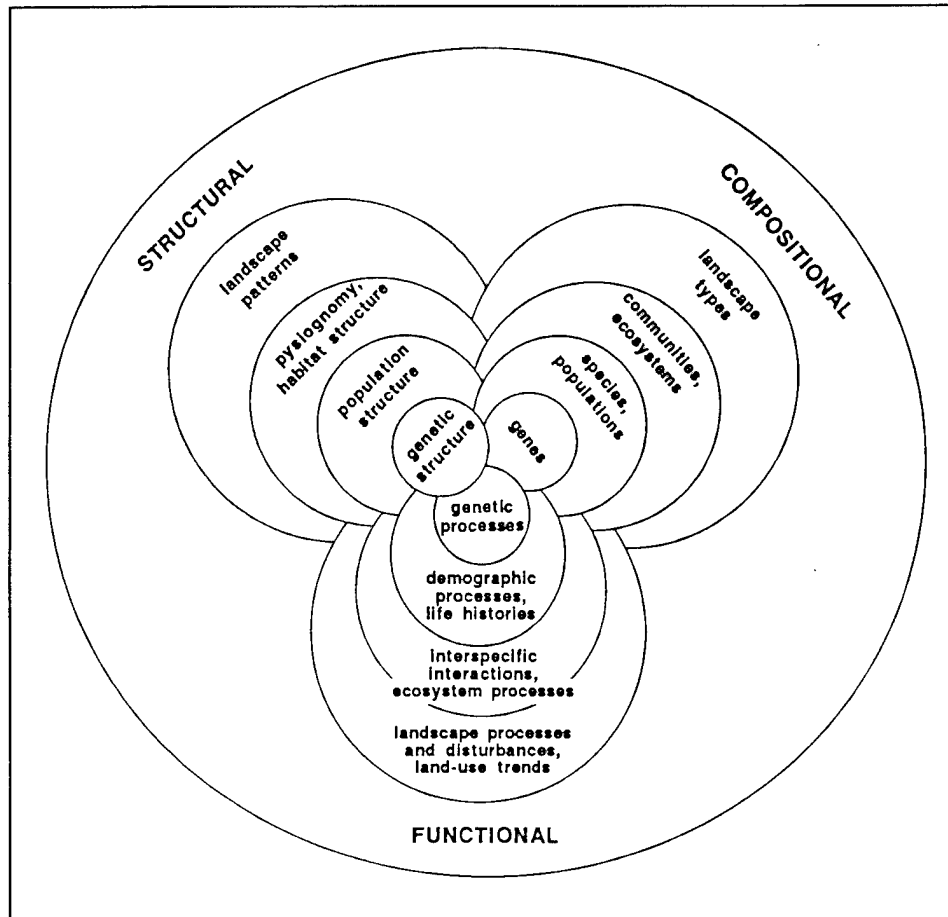


Figure 9. Compositional, structural, and functional biodiversity, shown as interconnected spheres, each encompassing multiple levels of organization (from Noss (1990:357); reprinted by permission of the Society of Conservation Biology and Blackwell Scientific Publications, Inc.)

Loss of genetic variation because of genetic drift in a small, isolated population results in both long- and short-term costs. Long-term costs include decreased raw material for evolutionary adaptations to changing environments, which may affect long-term survival of life on Earth (O'Connell and Noss 1992). In the short-term, decreased genetic variation because of genetic drift manifests itself as increased homozygosity (or inbreeding depression). As populations become smaller and more isolated, inbreeding because of genetic drift can reduce individual survival and productivity (Gaines et al. 1992). Ultimately, extinction probability can be increased by inbreeding effects in small, isolated populations (Mills and Smouse 1994).

Table 10
Indicator Variables and Tools/Techniques for Inventorying, Monitoring, and Assessing Terrestrial Biodiversity
at Four Levels of Organization, Including Compositional, Structural, and Functional Components¹

| Level of Organization | Indicator Variables | | | Inventory and Monitoring Tools |
|-----------------------|--|--|---|---|
| | Composition | Structure | Function | |
| Regional-landscape | Identity, distribution, richness, and proportions of patch (habitat) types and multipatch landscape types; collective patterns of species distributions (richness, endemism) | Heterogeneity; connectivity; spatial linkage; patchiness; porosity; contrast; grain size; fragmentation; configuration; juxtaposition; patch-size-frequency distribution; perimeter-area ratio; pattern-of-habitat-layer distribution | Disturbance processes (areal extent, frequency or return interval, rotation period, predictability, intensity, severity, seasonality); nutrient cycling rates; energy flow rates; patch persistence and turnover rates; rates of erosion and geomorphic and hydrologic processes; human land-use trends | Aerial photography (satellite and conventional aircraft) and other remote sensing data; Geographic Information Systems (GIS) technology; time series analysis; spatial statistics; mathematical indices (of pattern, heterogeneity, connectivity, layering, diversity, edge, morphology, autocorrelation, fractal dimension) |
| Community-ecosystem | Identity, relative abundance, frequency, richness, evenness, and diversity of species and guilds; proportions of endemic, exotic, threatened, and endangered species; dominance-diversity curves; life-form proportions; similarity coefficients; C4-C3 plant species ratios | Substrate and soil variables; slope and aspect; vegetation biomass and physiognomy; foliage density and layering; horizontal patchiness; canopy openness and gap proportions; abundance, density, and distribution of key physical features (e.g., cliffs, outcrops, sinks) and structural elements (snags, down logs); water and resource (e.g., mast) availability; snow cover | Biomass and resource productivity; herbivory, parasitism, and predation rates; colonization and local extinction rates; patch dynamics (fine-scale disturbance processes), nutrient cycling rates; human intrusion rates and intensities | Aerial photographs and other remote sensing data; ground-level photo stations; time series analysis; physical habitat measures and resource inventories; habitat suitability indices (HSI, multispecies); observations, censuses and inventories, captures, and other sampling methodologies; mathematical indices (e.g., of diversity, heterogeneity, layering dispersion, biotic integrity) |
| (Continued) | | | | |

¹ Source: Noss (1990:359); reprinted by permission of the Society for Conservation Biology and Blackwell Scientific Publications, Inc.

Table 10 (Concluded)

| Level of Organization | Indicator Variables | | | Inventory and Monitoring Tools |
|-----------------------|--|---|--|--|
| | Composition | Structure | Function | |
| Population-species | Absolute or relative abundance; frequency; importance or cover value; biomass; density | Dispersion (microdistribution); range (macrodistribution); population structure (sex ratio, age ratio); habitat variables (see community-ecosystem structure, above); within-individual morphological variability | Demographic processes (fertility, recruitment rate, survivorship, mortality); metapopulation dynamics; population fluctuations; physiology; life history; phenology; growth rate (of individuals); acclimation; adaptation | Censuses (observations, counts, captures, signs, radio-tracking); remote sensing; habitat suitability index (HSI); species-habitat modeling; population viability analysis |
| Genetic | Allelic diversity; presence of particular rare alleles, deleterious recessives, or karyotypic variants | Censuses and effective population size; heterozygosity; chromosomal or phenotypic polymorphism; generation overlap; heritability | Inbreeding depression; outbreeding rate; rate of genetic drift; gene flow; mutation rate; selection intensity | Electrophoresis; karyotypic analysis; DNA sequencing; offspring-parent regression; sib analysis; morphological analysis |

Reestablishing genetic diversity after it has been lost or reduced is not an easy task. Captive breeding and reintroduction is one approach to the problem, but it is expensive and techniques must be worked out for individual species (Ralls and Ballou 1992). Furthermore, problems arise because individuals used to start a captive population often contain only a small fraction of the total genetic variation of the parental population or species. This is known as the founder effect (Mayr 1970). However, genetic variation can be maintained or restored if enough viable founders are acquired (Ralls and Ballou 1992) and a positive population growth rate can be maintained (Vrijenhoek 1985). This implies that for reintroduction programs, population viability could be hampered by habitat loss or human conflicts. The current rate of habitat loss and degradation may prove to be an important limiting factor in reintroduction efforts. Obviously, the best approach is to protect natural communities and populations before they reach the point where captive breeding and reintroduction is the only alternative.

Species diversity

Species diversity refers to the number and variety of organisms in a given area (OTA 1987; Cairns and Lackey 1992). It is often described as (a) the total number of species in a community (i.e., species richness) or (b) the dual concept of diversity, which combines species richness and the relative abundance or evenness of species (Peet 1974; Westman 1990). For example, a community with an even number of individuals among species is considered more diverse than a community with a similar number of species but with a lower evenness (i.e., where some species are very abundant but others are rare). However, there is more to biodiversity conservation than species richness or diversity. Species composition is also an important consideration. For example, some species are always relatively low in number (e.g., large carnivores such as the Florida panther), but their mere presence may indicate good system integrity. Furthermore, resource managers must be aware of species composition and life-history patterns because human activities, including management actions, affect some species more than others (Noss 1983). For example, the introduction or invasion of nonnative species (e.g., purple loosestrife, water hyacinth, carp, and brown-headed cowbirds) can displace native biota and alter the composition and functions of a natural community (Westman 1990).

Certain species are known to have particularly strong interactions (*sensu* Mills, Soulé, and Doak 1993) and can have major effects upon species diversity. For example, within wetland ecosystems, species such as muskrats, nutria, beaver, and moose can dramatically alter vegetative structure, which subsequently influences the occurrence of other species (Harris 1988). Likewise, loss of species that play important functional roles (e.g., many decomposer microorganisms and ground-litter invertebrates) may negatively affect species diversity (Westman 1985, 1990). These species are often called “keystones” or “critical-links” (Paine 1969; O’Connell and Noss 1992). These terms have problems, however, because they imply a rigid species-specific property. In reality, interaction strengths vary spatially and temporally (i.e., a species may play a critical role in one wetland system, but the same species may play a less dramatic role in another wetland system). Hence, it is the biotic interactions and

their strengths within a particular system that are important, not necessarily the presence or absence of a species classified as “keystone” (Mills, Soulé, and Doak 1993; also see Landres, Verner, and Thomas 1988).

Species can also be characterized by level of diversity (Whittaker 1972) and vulnerability to environmental impacts. For example, Harris (1988:679) described the direct consequences of habitat fragmentation on wetland biodiversity:

- a. *Loss of large, wide-ranging species* (γ species), especially top carnivores or otherwise threatening forms (e.g., bears). Cursorial forms, which are vulnerable to automobile collisions, and aquatic migratory forms (e.g., fish, manatees), which are vulnerable to obstacles to migration, are particularly sensitive.
- b. *Loss of area-sensitive or interior species* (α species) that only reproduce in the interior of large tracts of wetlands and are therefore vulnerable to reduction in size of the individual component wetlands as well as reduction in total wetland acreage.
- c. *Loss of genetic integrity from within species or populations* that inhabit areas too small for a viable population of individuals. This is especially important for large, wide-ranging carnivores or raptors that are territorial and require areas proportional to population number (i.e., are not amenable to population packing).
- d. *Increase in abundance of habitat generalists characteristic of disturbed environments* (β species). Often these species serve as competitors (e.g., starlings), predators (e.g., crows and raccoons), or parasites (e.g., brown-headed cowbirds) on native species and accelerate their demise.

Ecosystem diversity

An ecosystem is “a community of organisms and their physical environment interacting as an ecological unit” (Lincoln, Boxshall, and Clark 1982:75). It is at the ecosystem level that biodiversity develops and is maintained (Landres 1992). Conversely, most efforts to conserve biodiversity have focused on species, subspecies, and populations (Franklin 1993). An ecosystem approach to biodiversity management offers important benefits that cannot be achieved at the species level. For example, an ecosystem-based approach is the only way to conserve the millions of species that constitute biodiversity—including the multitude of smaller, inconspicuous organisms that perform critical ecosystem functions (Szaro and Rinne 1988; Franklin 1993). Most of these smaller taxa are unknown and may never be known in a taxonomic sense. Hence, they will be conserved only if their ecosystems are conserved (Franklin 1993).

Managing ecosystems to maintain biodiversity is a difficult task because ecosystems are dynamic (i.e., their composition, structure, and functions change over time) and, thus, are subjectively defined (Landres 1992; O’Connell and

Noss 1992). Moreover, legislative mandates and practical knowledge to effectively manage many ecosystems are lacking (Clark et al. 1991). For example, little is known about long-term processes that sustain ecosystems, nor are the factors affecting temporal and spatial variation at a particular site completely understood (Landres 1992). Finally, integrated management of ecosystems is difficult because agencies charged with managing these areas have different agendas, responsibilities, and views of the management problem (Cairns 1990). Despite these shortcomings, an ecosystem-based approach is vital to the maintenance of biodiversity (Scott et al. 1988; Samson 1992; Landres 1992; Franklin 1993). Hence, ecosystem integrity must be a primary goal in land-use planning, and "the interconnectedness of ecosystems must be a fundamental concept in any management prescription" (Williams and Rinne 1992:5).

Landscape diversity

Cairns and Lackey (1992:7) defined landscape diversity as "the spatial heterogeneity of the various land-uses and ecosystems within a larger region measuring from 100 to 10,000 km²." However, there is no universally accepted definition of landscape. For example, some researchers and managers equate landscape management with watershed management. In either case, the focus of management is on large areas containing multiple ecosystems and land-use types.

Much of the temperate landscape has been altered by human activities such as agriculture, timber harvesting, livestock grazing, urbanization, and mining (Ratti and Scott 1991; Franklin 1993). Consequently, many landscapes are now dominated by a mixture of seminatural and domesticated lands (commonly termed the landscape matrix). For example, private agricultural land, excluding rangeland, accounts for nearly 29 percent of the total surface acres in the conterminous United States (U.S. Department of Agriculture 1988). In contrast, only about 3 percent of the world's land surface is protected and managed for natural values (Scott et al. 1993). Obviously, protected areas alone can no longer be relied upon to maintain biodiversity (Grumbine 1990). Consideration must also be given to the overall health of the landscape matrix (Noss 1983; Noss and Harris 1986; Hudson 1991; Harrison 1992; Martin 1992; Franklin 1993). For example, reducing environmental impacts, restoring natural habitats and landscape linkages, and protecting vital ecosystems (e.g., wetland and riparian) could restore a substantial portion of the biological diversity traditionally found in these human-dominated areas (Ratti and Scott 1991; Pimentel et al. 1992).

A healthy landscape matrix helps maintain biodiversity by (a) providing habitats across a wide array of spatial scales, (b) buffering reserve areas and increasing their effectiveness, and (c) providing for connectivity in the landscape, including movement of organisms between reserves (Franklin 1993). Although most scientists and resource managers agree on the relative importance of a healthy landscape matrix, there is much debate and confusion over the application of ecological theory (e.g., metapopulation dynamics, island biogeography, population viability, and edge effect) to landscape management and the design of nature reserves (see Simberloff and Abele 1982; Soulé and

Simberloff 1986; Reese and Ratti 1988; Murphy 1989; Martin 1992; Doak and Mills 1994). Resource personnel should be cognizant of these complex issues but avoid making or applying generalizations that may be of little or no practical use to specific conservation problems (Doak and Mills 1994). Hence, such generalizations were avoided in the discussion of landscape diversity. However, readers are urged to consult cited references for more detailed information on ecological theories and their possible application to landscape management.

Ecological and evolutionary processes

Long-term integrity and sustainability of ecosystems cannot be maintained without careful attention to ecological processes such as primary and secondary production, decomposition, energy and water flow, nutrient cycling, natural disturbances, succession, and interactions among species (Noss 1990; O'Connell and Noss 1992; Williams and Rinne 1992; Landres 1992). Evolutionary processes such as mutation, gene flow, geographic isolation, and hybridization also play a critical role in the preservation of present and future biodiversity (O'Connell and Noss 1992). Biodiversity, at all levels, both supports and depends on these ecosystem processes. For example, microbial populations play a critical role in chemical transformations within wetland soils, and chemical transformations enable plants to more easily assimilate nutrients such as nitrogen. Plants, in turn, convert sunlight into chemical energy (i.e., photosynthesis). This leads to a flow of energy and cycling of nutrients, which ultimately supports the diversity of life associated with wetland ecosystems (Weller 1987).

Value of Biodiversity

Biodiversity is valued for the many benefits it provides to humanity (Table 11). Organisms used to satisfy human needs are perhaps the most tangible and easily valued benefit. For example, the economic value of commercially harvested species can be established by cost-benefit analyses, and willingness-to-pay methods can be used to assess species with high esthetic, interest, or rarity values (Ehrlich and Ehrlich 1992). However, such analyses provide only partial values for species, i.e., standard analyses do not assess a species role in the food chain, their potential value to future generations, or the value of biodiversity taken in aggregate (Ehrlich and Ehrlich 1992).

Although more difficult to value, these less tangible aspects of biodiversity are equally important to humans. For example, biodiversity supports ecosystem services (primary production, decomposition, nutrient cycling, etc.) that are essential to human survival (OTA 1987). Clearly, a way must be found to address these benefits in the market system. "After all, a market system can hardly function to the ultimate benefit of humanity if it must classify the capacity for Earth to support life as an externality that cannot be properly internalized" (Ehrlich and Ehrlich 1992:226).

| Table 11 Examples of Benefits from Ecosystem, Species, and Genetic Diversity ¹ | | | | |
|---|--|---|--|--|
| Ecological Processes | Research | Cultural Heritage | Recreation and Tourism | Agriculture and Harvested Resources |
| Ecosystem Diversity | | | | |
| Maintenance of productivity; buffering environmental changes; watershed and coastal protection | Natural research areas; sites for baseline monitoring (e.g., Serengeti National Park, Zambesi Teak Forest) | Sacred mountains and groves; historic landmarks and landscapes (e.g., Mount Fuji; Voyageurs Park, Minnesota) | 700 to 800 million visitors per year to U.S. State and national parks; 250,000 to 500,000 visitors per year to mangrove forests in Venezuela | Rangelands for livestock production; habitats for wild pollinators and pest enemies (e.g., saving \$40 to \$60 per acre for grape growers) |
| Species Diversity | | | | |
| Role of plants and animals in forest regeneration, grassland production, and marine nutrient cycling; mobile links; natural fuel stations | Models for research on human diseases and drug synthesis (e.g., bristlecone pine, desert pupfish, medicinal leeches) | National symbols (e.g., bald eagle); totems; objects of civic pride (e.g., Port Orford cedar, bowhead whale, <i>Ficus religiosa</i>) | 95 million people feed, observe, and/or photograph wildlife each year; 54 million fish; 19 million hunt | Commercial logging, fishing, and other harvesting industries (\$27 billion/year in U.S.); new crops (e.g., kiwi fruit, red deer, catfish, and loblolly pine) |
| Genetic Diversity | | | | |
| Raw material of evolution required for survival and adaptation of species and populations | Fruit flies in genetics, corn in inheritance, and <i>Nicotiana</i> in virus studies | Breeds and cultivars of ceremonial, historic, esthetic, or culinary value (e.g., Texas loghorn cattle, rice festivals [Nepal]) | 100,000 visitors per year to Rare Breeds Survival Trust in the United Kingdom | Required to avoid negative selection and enhancement programs; pest and disease resistance alleles |
| ¹ Source: Office of Technology Assessment (1987:38). | | | | |

Status of Biodiversity

Taxonomists have classified about 1.7 million species, most of which are plants and insects (Figure 10). However, few species have been subject to detailed biological studies. Consequently, information on life-history and habitat requirements is lacking for most species, especially in poorly studied groups (e.g., invertebrates and microorganisms) and habitats (e.g., coral reefs, deep-sea floors, soils, forest canopies, and aquatic systems) (National Science Board 1989; NRC 1992b). Moreover, the number of classified species is only a small fraction of actual-species diversity, which is estimated to be between 10 million and 100 million living species (OTA 1987).

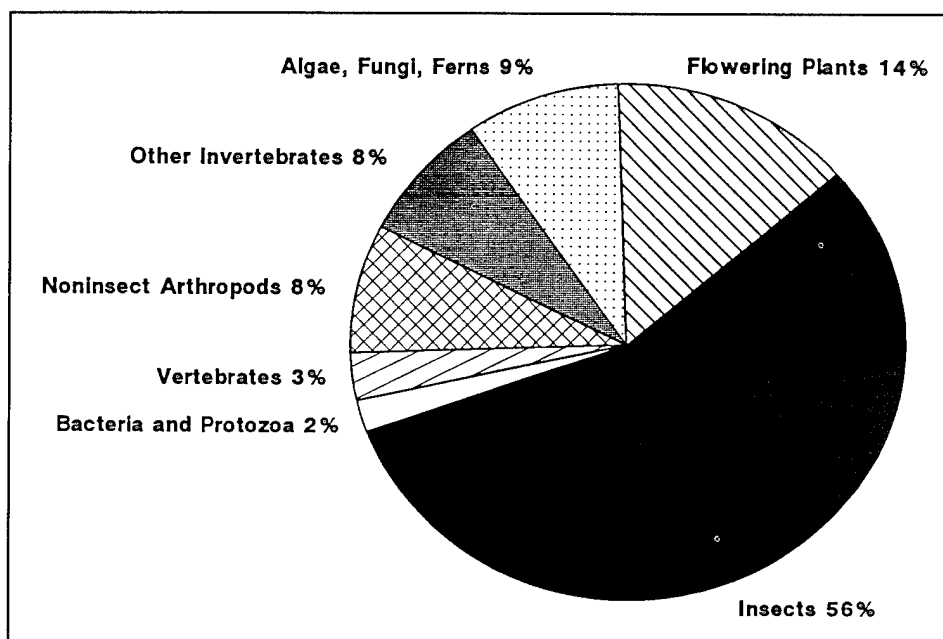


Figure 10. Categories of species that are classified and included in biodiversity (from the Office of Technology Assessment (1987:64))

Although precise estimates of the number and rate of species loss are impossible (simply because the exact number of living species is not even known to within an order of magnitude), there is no doubt that extinction is proceeding much faster than it did before 1800 (Wilson 1988). Furthermore, if habitat fragmentation and simplification continue, the current rate of extinction seems destined to approach that of the great natural catastrophes at the end of the Paleozoic and Mesozoic eras (Wilson 1988). Species diversity has fluctuated through geologic time as a result of evolution and extinction (Ehrlich and Wilson 1991; NRC 1992b:17; Jablonski 1991). However, there are important differences between current environmental changes affecting biodiversity and natural evolutionary changes. Today's environmental changes are extensive and occurring so rapidly that many species may not have sufficient time for an adaptive-evolutionary response (NRC 1992b:17). Consequently, contemporary extinction rates are estimated to be 1,000 to 10,000 times higher than the normal background extinction rates expected in the absence of human influences

(Wilson 1988). "Even conservative estimates of species-loss rates suggest that unless current trends are reversed, more than one-quarter of the Earth's species may vanish in the next 50 years" (NRC 1992b:18).

The impact on ecosystem services is no less alarming. For example, almost 40 percent of all potential net primary production (NPP; the energy fixed by photosynthesis) on land is directly consumed, diverted, or forgone because of human activities (Ehrlich and Ehrlich 1992). The amount of terrestrial NPP available to accommodate further expansion of the human population and its mobilization of resources is limited, especially if the human population doubles in the next half-century as predicted (Ehrlich and Ehrlich 1990, 1992). Biodiversity conservation is clearly at a very critical juncture.

The causes of biodiversity loss include (a) large-scale clearing and burning of forests; (b) intentional and incidental overharvesting of plants and animals; (c) use of pesticides and herbicides; (d) degradation of wetlands and riparian areas; (e) air pollution; (f) habitat alteration, fragmentation, and simplification, including the extensive conversion of wildlands to agricultural and urban uses; (g) introduction of exotic species; and (h) stress from global atmospheric change (McNeely et al. 1990; Cairns and Lackey 1992). However, some traditional habitat management practices may even promote biodiversity loss (Martin 1992; Cairns and Lackey 1992). For example, creation of edge habitat is a common strategy to benefit wildlife populations (Leopold 1933:132; Yoakum et al. 1980; Robinson and Bolen 1984; Reese and Ratti 1988). However, excessive edge habitat can decrease species diversity and alter species composition within the community (Ambuel and Temple 1983; Laudenslayer 1986; Temple 1986; Temple and Wilcox 1986; Reese and Ratti 1988; Guthery and Bingham 1992). Furthermore, edge habitats may act as an ecological trap; for example, some passerine species will be attracted to the vegetative diversity of edge habitats but may experience increased losses to predation and brood parasitism (Gates and Gysel 1978; Chasko and Gates 1982; Reese and Ratti 1988). More research is needed to assess impacts of edge creation, location, and manipulation on species diversity and population viability (Yahner and Wright 1985; Ratti and Reese 1988; Reese and Ratti 1988).

The ultimate cause of biodiversity loss, however, is the combination of human population growth, technological advances, and increased demands for goods and services (Ehrlich and Ehrlich 1990). This is a global problem that involves both technologically advanced and less developed countries. Although recent concern over the loss of biodiversity was prompted by the accelerated rate of deforestation in tropical rain forests, global degradation of other natural ecosystems and habitats is equally important.

Wetland and Deepwater Habitats

Man has altered a substantial portion of the landscape and water bodies in the United States. For example, <2 percent of the 5,200,000 km of streams in the conterminous United States remain in high-quality condition (Benke 1990; Williams and Rinne 1992); only 25 to 46 percent of riparian communities remain

in near-natural conditions (Hughes and Noss 1992); and nearly 50 percent of our nation's wetlands have been lost (Dahl 1990). Consequently, species loss may be as great or greater in temperate regions than in the tropics (Hughes and Noss 1992). Perhaps the greatest declines in biodiversity of wetlands and deepwater habitats have occurred in western North America, where there is intense demand for water by humans (Moyle and Williams 1990). However, the decline in biodiversity is equally alarming in other geographic areas. For example, 40 to 50 percent of the freshwater snails in the southeastern United States are extinct or near extinction (McNeely et al. 1990). Hughes and Noss (1992) described other examples of biodiversity loss at the genetic, species, ecosystem, and landscape levels of organization.

Wetland and deepwater habitats should be an important part of biodiversity-conservation efforts. Wetlands not only contribute ecological functions (e.g., store surface water, recharge groundwater, transform nutrients) but provide habitat for fish, birds, and other wildlife (Council on Environmental Quality (CEQ) 1989; NRC 1992a). For example, wetlands serve as nurseries and feeding areas for fish and shellfish, and support about one-third of North American bird species (CEQ 1989). Furthermore, many endangered and threatened species use wetlands during some part of their life cycle (Table 12; also see Niering 1988). Riverine habitats and palustrine wetlands are especially vital to endangered and threatened species (Table 13).

Table 12
Wetland-Dependent Taxa Federally Listed as Threatened or Endangered¹

| Taxa | Threatened ² | Endangered ² | Total ² | Total Taxa Listed | % Wetland Dependent |
|-------------|-------------------------|-------------------------|--------------------|-------------------|---------------------|
| Plants | 19 | 35 | 54 | 281 | 19 |
| Animals | 57 | 145 | 202 | 314 | 64 |
| Mammals | 3 | 10 | 13 | 41 | 32 |
| Birds | 5 | 23 | 28 | 64 | 44 |
| Fishes | 33 | 51 | 84 | 84 | 100 |
| Reptiles | 3 | 2 | 5 | 22 | 23 |
| Amphibians | 4 | 10 | 14 | 14 | 100 |
| Insects | 5 | 3 | 8 | 21 | 38 |
| Arachnids | 0 | 0 | 0 | 3 | 0 |
| Crustaceans | 0 | 5 | 5 | 10 | 50 |
| Snails | 2 | 3 | 5 | 13 | 38 |
| Bivalves | 2 | 38 | 40 | 42 | 95* |
| Total | 76 | 180 | 256 | 595 | 43 |

Note: * = Did not total to 100 percent because of incomplete data.

¹ Source: Based on data from Feierabend (1992).

² Number of wetland-dependent species (i.e., any species that use wetlands or other aquatic habitats at some point during their life cycle; includes both obligate and facultative wetland species).

Table 13
Number of Threatened or Endangered Species by Wetland-Habitat Type¹

| Habitat ² | Threatened ³ | Endangered ³ | Total ⁴ | % of Total Dependent Species ⁵ | % of Total Species Listed ⁶ |
|----------------------|-------------------------|-------------------------|--------------------|---|--|
| Palustrine | 25 | 70 | 95 | 37 | 16 |
| Lacustrine | 7 | 29 | 36 | 14 | 6 |
| Estuarine | 7 | 28 | 35 | 14 | 6 |
| Riverine | 45 | 110 | 155 | 61 | 26 |
| Marine | 6 | 12 | 18 | 7 | 3 |
| Vernal Pools | 3 | 2 | 5 | 2 | 1 |

¹ Source: Based on data from Feierabend (1992).

² Wetlands and deepwater habitats; after Cowardin et al. (1979).

³ Number of wetland-dependent species (i.e., any species that use wetlands or other aquatic habitats at some point during their life cycle; includes both obligate and facultative wetland species).

⁴ Totals are not mutually exclusive, i.e., some species use more than one habitat type (e.g., bald eagles use riverine, estuarine, lacustrine, and palustrine wetlands).

⁵ Feierabend (1992) classified 256 threatened and endangered species as wetland dependent.

⁶ The U.S. Fish and Wildlife Service listed a total of 595 plant and animal species as threatened or endangered in 1991 (Feierabend 1992).

Several attributes of wetlands are particularly important for maintaining biodiversity (Table 14). However, species diversity at a particular site may be affected by many factors (e.g., Figure 11). Interactions among adjacent ecological units is another factor to consider. These interactions can influence community composition and population dynamics (Burger 1985; Weller 1987; Risser 1990). For example, Karr and Schlosser (1978) reported that removal of near-stream vegetation in upstream areas significantly reduced invertebrate and fish production through loss of allochthonous-energy inputs into adjacent streams. Invertebrates are an important food source for fish in riverine habitats and for breeding and wintering waterbirds in wetland habitats (Fredrickson and Reid 1986; Fredrickson and Batema 1992). Thus, managers must be cognizant of activities that could negatively affect invertebrate production.

Vegetation structure and water depth are key factors in wetland management (Fredrickson and Taylor 1982). These factors influence habitat selection and community structure (especially avian) in both forested wetlands (Szaro 1980; Swift, Larson, and DeGraff 1984) and emergent wetlands (Weller and Spatcher 1965; Burger 1985; Weller 1987). The greatest diversity of organisms is usually found in large-wetland complexes with the following conditions: (a) a mix of habitats ranging from open water and mudflats to dense rank vegetation, (b) a good interspersed of open water (50 to 70 percent) and cover, and (c) relatively shallow water levels (i.e., <45 cm) (Weller and Spatcher 1965; Fredrickson and Reid 1986; Reid 1993). Individual sites may vary, however. For example, ideal water depths for a given area will depend on primary-wildlife users and on the ability to control water levels (see Fredrickson and Taylor 1982; Fredrickson and Reid 1986; Fredrickson and Batema 1992; Payne 1992:177-181).

Table 14
Wetland Attributes That Assist in the Maintenance of Biodiversity¹

| Attribute | Description |
|---|---|
| Persistence of habitat | Important for mating, nesting, and protection from predators during extreme environmental conditions. |
| Resilience | Ability to recover from natural or human disturbances (e.g., environmental extremes such as tidal closure and drought), often conferred through marsh soils. |
| Ability to maintain plant populations | Regions with high environmental variability need refuges for long-term maintenance of populations and to ensure resilience (ability to recover rapidly) following extreme events. |
| Resistance to invasive species | The continual threats of disturbance to topography and hydrology lead to the need for constructed wetlands to resist invasive species (exotic to the region or alien to the habitat). |
| Nutrient transformations | Microbial and chemical processes control the concentrations of nutrients and other compounds and facilitate the biogeochemical cycling of nutrients and energy flow. |
| Productivity | Wetland production is important to both aquatic- and terrestrial-food webs. |
| Travel corridors | Wetlands (especially riverine wetlands) serve as corridors for large, far-ranging species such as the Florida panther and black bear, as well as wetland-dependent species such as amphibians. Some areas may also act as migration corridors for neotropical migrants. On a smaller scale, wetland corridors may be important for waterfowl-brood movements. |
| ¹ Source: After the National Research Council (1992a:265-267) and Harris (1988). | |

Managers must be flexible and understand the needs of the local flora and fauna. For example, individual species often require multiple-wetland types to complete their annual cycle. Thus, the pattern and composition of a wetland complex can strongly influence species richness (Laubhan and Fredrickson 1993). Large complexes of wetlands, which usually have a good interspersion and juxtaposition of habitats, typically have more species of breeding birds than small, isolated wetlands (Brown and Dinsmore 1986; Gibbs et al. 1991). Similarly, loss of wetlands within a community may reduce use of remaining wetlands (Flake 1979) or at least reduce reproductive efficiency (Rotella and Ratti 1992). The optimal pattern and composition of a wetland complex depend on life-history requirements and mobility of the species known to occur in the area, with the least mobile species often dictating the correct pattern (Laubhan and Fredrickson 1993).

Other landscape features can also influence species distribution and abundance (see Forman and Gordon 1981; Urban, O'Neill, and Shugart 1987; Barret and Bohlen 1991). For example, Parcels and Dunstan (1993) described the importance of buffer areas and habitat corridors in managing wetland complexes for biodiversity. A manager should also consider larger scale factors. For example, migratory species depend on the availability of geographically distinct habitats (i.e., staging, migration, breeding, wintering, and molting areas) to complete their annual cycle (Myers et al. 1987). Habitat loss and degradation in any portion of their geographic range can influence population dynamics.

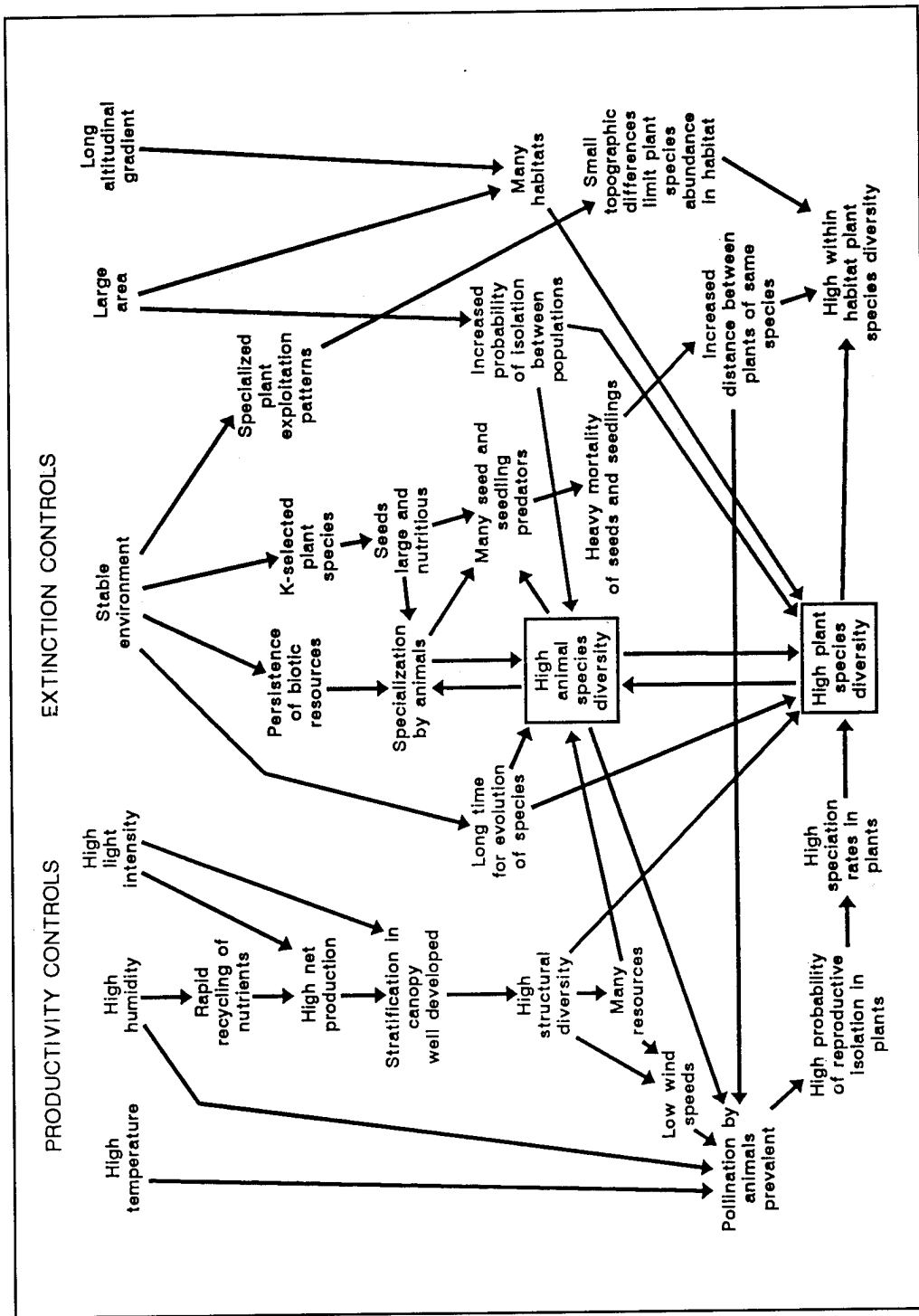


Figure 11. Summary of processes leading to high diversity in terrestrial ecosystems (from Colvinaux (1986:674); copyright 1986 by John Wiley and Sons, Inc., reprinted with permission)

Wetland and deepwater ecosystems play an important role in the maintenance of biodiversity. Nevertheless, they have received little attention in national forums (Blockstein 1992). The 1986 National Forum on Biodiversity had no speakers on freshwater biodiversity (except in terms of ecological restoration) and only two on marine biodiversity (Wilson 1988). Some information is available on nongame species associated with wetlands (Fredrickson and Reid 1986; Svedarsky 1992; Helmers 1992; Reid 1993; Laubhan and Fredrickson 1993); however, more information is needed to effectively manage wetlands in a truly integrated manner (i.e., considering all species within the community). Nevertheless, implementing conservation strategies now must be attempted because biodiversity (especially native fauna) has declined dramatically in wetland and deepwater habitats (Moyle and Williams 1990; Hughes and Noss 1992; Upton 1992) and is rapidly approaching a point of critical concern (Blockstein 1992).

Conservation Strategies

The traditional response to declining biodiversity has focused on saving selected species through the Endangered Species Act (ESA). Despite the ESA, the number of endangered, threatened, and candidate species continues to grow. Furthermore, natural communities and ecosystems that support endangered and threatened species (along with myriad inconspicuous species) continue to deteriorate (Scott, Csuti, and Caicco 1991). The ESA remains an important biopolitical tool; however, it alone will not protect the full range of biodiversity. Consequently, many scientists have questioned the validity of this species-by-species approach to biodiversity conservation (e.g., Hutto, Reel, and Landres 1987; Scott et al. 1987; Scott, Csuti, and Caicco 1991; Noss 1991; Pitelka 1981; Meese 1989; Cairns and Lackey 1992).

Emphasis on sustainable ecosystems and biodiversity conservation represents a significant step beyond the endangered-species approach (Noss 1991; Scott, Csuti, and Caicco 1991; Samson 1992). For wetland managers, this means emphasizing community structure, productivity, and long-term integrity of the wetland complex and landscape matrix. However, developing standards for land management that are consistent with the goals of conserving biodiversity is not a simple task (Murphy 1989; Westman 1990), especially for private-land management (O'Connell and Noss 1992). O'Connell and Noss (1992:438) offered the following advice:

...responsible management does not preclude habitat alteration or even loss of certain species from a property. From a regional and global perspective, mitigation and restoration become tools allowing utilization of land and resources to proceed while minimizing or compensating for negative impacts on biodiversity.

However, there is much to be learned about restoring or creating functional attributes of complex ecosystems (Cairns 1988; Jordan 1988), including wetlands (Zedler 1988; D'Avanzo 1990; Zedler and Weller 1990; Weller 1990). Nevertheless, ecological restoration will become an increasingly important

strategy because too many ecosystems and too much of the landscape have been significantly altered. Reversing the process of biotic impoverishment by simply setting aside and maintaining small, scattered preserves in the remaining undisturbed ecosystems will no longer suffice (Ehrlich 1988; Grumbine 1990; Scott et al. 1993). A combination of strategies are needed: (a) restoration of degraded ecosystems (Cairns 1988; Jordan 1988; Zedler and Weller 1990; Hunter 1991; NRC 1992a; Parcells and Dunstan 1993), (b) creation of new or improved habitats (Duebber et al. 1981; Martin and Marcy 1989; Weller 1990; Roberts 1991; Payne 1992; Hammer 1992), (c) captive breeding and reintroduction programs (OTA 1987; Ralls and Ballou 1992; Gaines et al. 1992), (d) protection and management of the landscape matrix (Barrett and Bohlen 1991) and habitat corridors (Harris and Gallagher 1989; Harris and Atkins 1991; Beier and Loe 1992; but see Martin 1992), (e) new approaches to private-land management (Ratti and Scott 1991; O'Connell and Noss 1992), and (f) identification of unprotected biodiversity "hotspots" based on assessment of large geographic areas (Scott et al. 1993).

Conservation strategies, and associated complications, also apply to wetland management. Wetland management has slowly changed from a featured-species approach to a community-oriented approach that strives to provide benefits to a maximum number of species. Attempts to meet this new management challenge have been complicated by the loss and degradation of wetland complexes. Although conservation programs are now in place to encourage restoration/creation of lost wetlands, such programs often focus on the number of sites or total area restored or created. Little information exists on juxtaposition, wetland type, and hydrology of lost wetlands. Consequently, the type of wetlands being restored and created may be dissimilar from those being lost (Laubhan and Fredrickson 1993).

No single-wetland type provides the resources required by all species in a given period, nor does a single-wetland type provide the resources required for all stages in the annual cycle of a single species (Swanson, Krapu, and Serie 1979; Fredrickson and Reid 1986; Weller 1987, 1990; Fredrickson and Batema 1992; Reid 1993; Laubhan and Fredrickson 1993). Therefore, managers should strive to recreate and maintain a mosaic of wetland habitats that mirrors the unique biodiversity of the historic system, if possible (Parcells and Dunstan 1993). This can best be accomplished through a combined strategy of wetland protection, restoration, creation, and management (Ratti and Kadlec 1992; Laubhan and Fredrickson 1993).

7 Stewardship and Management

Until the middle of the twentieth century, wetland management and reclamation usually meant wetland drainage and conversion to more “valuable” uses such as agriculture and urban development. Although our national policy has changed to one of preservation, wetland management continues to mean different things to different disciplines and interest groups. In general, however, wetlands are managed for environmental protection, recreation and aesthetics, and production of renewable resources (Mitsch and Gosselink 1986:429).

Wetland management may involve acquisition and protection or more manipulative strategies such as structural and biological alterations. Furthermore, wetland management may focus on a specific goal or encompass a broad range of objectives. Recently, more emphasis has been placed on multipurpose management because efforts to maximize one objective are often incompatible with the attainment of others. The focus of this review is the management of wetlands for wildlife. Traditionally, this meant management of wetlands for waterfowl. Some wetlands continue to be managed primarily for waterfowl; however, most resource personnel now advocate broader management programs (Weller 1982; Mathisen 1985; Fredrickson and Reid 1986; Helmers 1992; Laubhan and Fredrickson 1993; Reid 1993).

The science of wetland management has moved from trial-and-error to development of technical skills and then to theory. Despite these advances, wetland management cannot be reduced to a series of cookbook procedures. However, the probability of success can be increased by understanding the natural patterns and processes of wetlands. For example, better decisions will be made if the manager understands the structure and dynamic nature of wetlands and relationships among wetlands and other ecological units in the landscape.

The Concept of Management

Protection and maintenance of habitats is a key component in wildlife management (Yoakum et al. 1980) for both game and nongame species (Hale, Best, and Clawson 1986). The continued loss, fragmentation, and general degradation of habitats underlies the significance of habitat programs. For wetland-associated species, the importance of habitat complexes cannot be overemphasized. The protection and maintenance of different wetland habitats

or wetland types, arranged in close juxtaposition, helps to maintain faunal and floral diversity typical of a region and avoids endangering some species while overproducing others (Weller 1987:71). This diversity of wetland habitats is important because a single-wetland type usually does not provide all resources needed by a group of species or even the resources necessary to complete all stages in the life cycle of one species. For example, Fredrickson and Reid (1986) described the importance of diverse habitats to wildlife taxa using southern-floodplain forests (Table 15) and glacial wetlands (Table 16).

Acquiring and preserving a balanced pattern of representatives of all wetland types in an area may not always be possible. In such cases, manipulation of remaining wetlands may be required to produce a diversity of wetland habitats (Weller 1987:71). However, each site must be carefully assessed for its management potential prior to undertaking expensive manipulations (Fredrickson and Reid 1986:69, 1988a; Payne 1992). This includes considering the site's potential to support wildlife, either through featured-species management (i.e., the traditional approach to wildlife management) or ecosystem management (i.e., a strategy for biodiversity conservation).

Featured-species and ecosystem management may appear to be diametric strategies; however, in some cases, little difference exists between the two approaches. For example, many species (including nongame and terrestrial) benefit from wetland-management activities directed at waterfowl, which have broad habitat requirements (Rakstad and Probst 1985; Payne 1992; Fredrickson and Batema 1992; Helmers 1992). The exception is when the featured or target species has special status (endangered or threatened) and a narrow niche (Payne 1992:2). However, management is a continuous process, and information on a species' biology and habitat requirements should be incorporated into the ongoing operation when it becomes available. This is an important point because very little is presently known about habitat requirements of many nongame species (Fredrickson and Reid 1986). In general, active management is undertaken only when (a) it provides the nucleus for improving a larger area of habitat; (b) it is the only way to provide a missing, essential factor; or (c) it restores habitat damaged or altered by human activity or catastrophic weather and when the system cannot be restored naturally in a reasonable amount of time (Payne 1992:1).

Wetland management typically involves the manipulation of landform and/or successional stages of plant communities. Vegetative pattern and structure are especially important because they strongly influence the animal community, and vice versa in some cases (e.g., muskrats and nutria). Management practices emulating natural disturbances (e.g., seasonal water-level fluctuations) are best (Weller 1982); however, this typically requires expensive water-control structures such as dikes, weirs, spillways, control gates, and pumps. Whether such structures should be added to a natural marsh is debatable, although the success of management may justify the effort (Weller 1987:74).

| Table 15 Habitat Types and Major Wildlife Taxa of Southern-Floodplain Forests¹ | | | | | |
|--|--------------------------|---|---|-------------------------------|---|
| Wetland Type | Flood Duration | Plant Foods | Animal Foods | Cover | Wildlife Taxa ² |
| Open-water | Usually 12 months | None | Fish | None | Turtles, grebes, fish, pelicans, diving ducks |
| Aquatic bed: Submergent | 11-12 months | Seeds, Browse | Fish, Insects, Snails | None | Turtles, fish, grebes, pelicans, diving ducks |
| Aquatic bed: Watershield | 10-12 months | Seeds, Browse | Fish, Insects, Snails | None | Turtles, frogs, fish, gallinules, diving ducks, wood ducks |
| Emergent bed: American lotus | 8-12 months | | | Roosting | Snakes, frogs, wood ducks |
| Emergent wetland: Moist-soil | Variable 1-12 months | Seeds, Tubers, Browse | Crayfish, Insects, Snails | Nesting, Roosting, Feeding | Waders, rails, frogs, snakes, swallows, marsh wrens, dabbling ducks, blackbirds |
| Scrub/shrub | 8-12 months | Seeds | Crayfish, Insects, Snails | Nesting, Roosting | Snakes, frogs, waders, dabbling ducks, wood ducks |
| Forested: Cypress-tupelo | 6-8 months (Dec-July) | Tupelo drupes | Crayfish | Nesting, Roosting | Fish, frogs |
| Forested: Overcup oak | 4-6 months (Dec-May) | Large acorns, Berries, Fruits, Samaras | Crayfish, Small crustaceans, Fingernail clams | Nesting, Roosting | Frogs, snakes, woodpeckers, prothonotary warblers, tanagers, dabbling ducks |
| Forested: Pin/nuttall oak | 1-6 months (Dec-May) | Small acorns, Berries, Fruits, Samaras | Crayfish, Spiders, Small crustaceans | Nesting, Roosting | Frogs, woodpeckers, red-shouldered hawks, dabbling ducks, wood ducks |
| Forested: Cherrybark/ willow oak | 1-3 months (Jan-Mar) | Small acorns, Berries, Fruits, Samaras | Spiders, Small crustaceans | Nesting, Roosting | Woodpeckers, red-shouldered hawks, dabbling ducks, wood ducks |

¹ Source: after Fredrickson and Reid (1986:68-69).

² Major groups of wildlife species (except for a few specific examples) using southern-floodplain habitats.

| Table 16 Habitat Types and Major Wildlife Taxa of Glacial Wetlands ¹ | | | | | | |
|--|----------------|--------------------------|---|-------------------|--|--|
| Wetland Type | Flood Duration | Plant Foods | Animal Foods | Cover | Wildlife Taxa ² | |
| Type I - ephemeral ponds | < 1 month | Seeds, Browse | Insects, Fairy shrimp | None | Shorebirds, dabbling ducks | |
| Type II - temporary ponds | < 1 month | Seeds, Browse | Insects, Fairy shrimp | None | Shorebirds, waders, dabbling ducks | |
| Type III - seasonal ponds and lakes | 2-4 months | Seeds, Browse, Tubers | Insects, Snails | Nesting, Roosting | Frogs, shorebirds, waders, rails, sedge wrens, dabbling ducks, diving ducks | |
| Type IV - semipermanent ponds and lakes | 12 months | Seeds, Browse | Insects, Snails, Crustaceans, Chironomids | Nesting, Roosting | Turtles, frogs, waders, grebes, rails, gallinules, pelicans, blackbirds, terns, dabbling ducks, diving ducks | |
| Type V - permanent ponds and lakes | 12 months | Seeds, Browse | Insects, Snails, Chironomids | Nesting, Roosting | Turtles, fish, diving ducks | |
| Type VI - alkali ponds and lakes | 6-12 months | Submergent seeds, Browse | Small crustaceans | None | Shorebirds, dabbling ducks, diving ducks | |
| Type VII - fen ponds | 12 months | Seeds | Snails | Nesting, Roosting | Blackbirds | |

¹ Source: after Fredrickson and Reid (1986:70-71).

² Major groups of wildlife species (except for a few specific examples) using glacial wetlands.

Other management practices that emulate natural disturbances include controlled grazing and the use of fire. Although these practices may appear to be drastic (e.g., a complete drawdown or a prescribed burn), they often are the most ecologically and economically sound methods available. However, care must be taken "to reduce excessive, often unnecessary, and overly artificial management programs..." (Weller 1987:72). This can best be accomplished by following Weller's (1982:949) general principles of wetland management for wildlife:

- a. System, rather than species management, results in widespread benefits to all plants and wildlife. Although there is some evidence of competition in birds, losses in production of game species due to high species richness of nongame species have not been demonstrated.
- b. Manipulation to produce early plant successional stages results in longer lasting benefits and creates diverse habitat niches. A marsh then proceeds through various phases with productivity of any one species being a dynamic component of the system. Methods producing long-term results are less expensive and more natural. Usually, this means use of natural tools for management.
- c. To maintain heterogeneity in wetland complexes, all marsh units in an area should not be managed in the same way at the same time, even with extreme climatic conditions. This permits local population shifts of wildlife to more optimal niches.
- d. Tools such as remote sensing offer exciting opportunities to enhance and document marsh management, but there is no substitute for the manager getting into the marsh.

The specific goals and practices used in wetland management depend on a complex of factors that influence composition and structure of the plant community. Factors such as topography, season, time of drawdown, type of drawdown, type of disturbance, time since disturbance, time since continuous flooding, long-term patterns of precipitation, and interactions among the seed bank are important considerations (Fredrickson and Reid 1986). The type of wetland system, its geographic location, role in supporting wildlife populations, and the ability to control water levels may also influence management objectives. General guidelines have been established for some wetland types; however, managers typically must adjust for regional and site variation.

Managed Wetland Types

Managed wetlands can be grouped into five types based on management objectives and techniques: (a) freshwater marshes (persistent-emergent wetlands and open-water ponds); (b) moist-soil impoundments managed for nonpersistent emergents; (c) greentree reservoirs; (d) tidal-estuarine wetlands (salt and brackish marshes); and (e) dredge-fill wetlands. The management of tidal and dredge-fill wetlands is not covered in this review, but detailed information can be found in other sources (Atlantic Waterfowl Council 1972; U.S. Army Engineer

Waterways Experimental Station 1978; Gordon et al. 1989; Weller 1990; and Payne 1992).

Freshwater marshes

Most northern and midlatitude marshes are managed for breeding populations of birds (especially waterfowl and shorebirds) and other wetland wildlife; whereas southern marshes, and some northern and midlatitude marshes, are commonly managed for migrating and wintering wildlife. In both cases, management usually favors persistent-emergent plants that provide food and structure (i.e., nest sites and cover) for wetland wildlife. On breeding areas, however, management usually focuses on habitat structure more than composition; that is, persistent-perennial plants such as cattail and bulrush are favored (Weller 1987; Kadlec and Smith 1992). Conversely, freshwater marshes important to migrating and wintering wildlife are managed mostly for food-producing plants, although structure (cover) remains important for animal-prey and nonmigratory organisms (Fredrickson and Reid 1986).

Most managers, however, strive to provide a diversity of plants of assorted life forms to serve numerous animals (Weller 1990). For example, nonpersistent-emergent plants (e.g., annuals) produce large seed-crops; persistent emergents (e.g., cattail and bulrush) provide structure for nest sites, and the tuberous bases are used by herbivores; and submergent plants provide food directly or serve as substrates for invertebrates, which are vital components in nutrient cycling and decomposition. Persistent vegetation also helps control bank erosion.

The best marsh complex contains a mix of three habitats: moist-soil or mudflats, shallow marsh, and deep marsh (Payne 1992:179). In addition, these wetlands typically have structure ranging from open water or mudflats to dense, rank vegetation (Fredrickson and Reid 1986). In many cases, the amount of open water and interspersed emergent vegetation is also important. Managers generally try to maintain 50- to 70-percent open water, although management objectives may vary depending on the primary-wildlife users and feasibility of vegetative manipulations. For example, wetlands managed mainly for refuge and resident wildlife usually contain more deep-marsh and shallow-marsh habitat (Bookhout, Bednarik, and Kroll 1989); wetlands important to migrating and wintering shorebirds usually have more mudflat or moist-soil habitat (Helmert 1992); and wetlands used mostly by dabbling ducks (e.g., mallard and blue-winged teal) and wading birds (e.g., herons and egrets) usually contain more shallow-water (≤ 25 cm) habitat (Fredrickson and Reid 1986). In most cases, however, managers try to promote high-species richness by providing a diversity of structure and food resources, including invertebrates and plants (Fredrickson and Reid 1986; Weller 1987; Payne 1992).

Management techniques vary according to the type of marsh and ability to control water levels. Natural techniques such as water-level manipulation are preferred but usually are restricted to wetlands with water-control structures (e.g., impoundments). Drawdowns every 2 to 4 years are best for areas managed

primarily for breeding waterfowl (Linde 1969); however, successful modifications of this practice have been reported. For example, a management scheme used successfully in Wisconsin is to conduct drawdowns 2 or 3 years in a row, then skip 1 or 2 years (Payne 1992:185). Another common practice is to expose about half the bottom for at least 3 months during the growing season every 2 or 3 years (Green, MacNamara, and Uhler 1964; Payne 1992).

In some cases, artificial methods such as structural modification and mechanical or chemical treatment of the vegetation may be required. These techniques are often used on wetlands without water-control structures, e.g., potholes, playa lakes, vernal pools, farm ponds, mines and gravel pits, beaver ponds, and riparian areas. However, nonstructural management also should be considered in these situations. For example, beaver ponds attract a wide variety of wildlife; but the typical beaver impoundment is constantly changing and evolving (Yoakum et al. 1980). Consequently, the focus of management should not be on maintenance of one pond, but on the rotation of favorable-habitat elements within the entire area of beaver influence (Yoakum et al. 1980). In some cases, however, active management may be possible and beneficial. Hair et al. (1979), Arner and Hepp (1989), and Ringleman (1991) reviewed the management of beaver-pond ecosystems to benefit waterfowl and other wildlife.

Adjacent uplands also need to be considered in the overall-management plan. Upland habitats are used by a wide diversity of species for nesting (e.g., dabbling ducks) and foraging. Upland areas are also important for soil conservation and chemical filtering. Furthermore, manipulations of uplands are often easier and more economical than the manipulation of wetland basins to enhance habitats (Fredrickson and Reid 1986). Management techniques include fire, grazing, and mowing, and the establishment of dense nesting cover (Payne 1992; Payne and Bryant 1993).

Moist-soil impoundments

Moist-soil impoundments are a special type of freshwater marsh in which water level can be carefully controlled through a system of dikes, gates or pumps, and landform contours. These areas are traditionally managed to provide foods for migrating and wintering waterfowl, although shorebirds and other wildlife also benefit (Fredrickson and Reid 1986). Because migratory waterfowl feed more on high-carbohydrate seeds and less on high-protein-animal foods in fall and winter, production by seed-producing plants is emphasized (Weller 1990). However, recent evidence suggested that tubers, rootlets, browse, invertebrates, and herpetofauna also are enhanced by moist soil management and provide a wide variety of foods for wildlife (Fredrickson and Reid 1986). Moist-soil management has recently become a popular management practice throughout much of the East and in some parts of the Midwest, both on State-owned wetlands and on national wildlife refuges (Reid et al. 1989).

Moist-soil management emulates natural-drawdown conditions of emergent wetlands that are seasonally flooded (Reid et al. 1989). Artificial drawdowns expose the soils, which allows germination of important seed-producing,

nonpersistent emergents such as millets, smartweeds, spikerushes, and other marsh-edge plants (Fredrickson and Taylor 1982). Drawdowns are normally performed early in the year to allow sufficient drying so that annual seeds will be produced (versus undesirable species); consequently, such wetlands generally are not breeding marshes for waterfowl (Weller 1990). The composition and structure of the plant community depend on timing of the drawdown and stage of wetland succession (Reid et al. 1989), as well as seasonal temperatures, rainfall, soil structure, soil type, seed-bank composition, duration and rate of drawdown, topography, and site variation (Fredrickson and Taylor 1982; Fredrickson and Reid 1986; Fredrickson 1991).

Moist-soil areas are usually reflooded in fall prior to arrival of waterfowl and other migrants. Managers attempt to maintain water depths between 15 to 46 cm. These shallow conditions provide the best foraging conditions for most species and reduce costs for flooding and development (Fredrickson and Reid 1986). Managers interested in attracting the greatest diversity of organisms usually strive for a mix of habitat conditions ranging from open water and mudflats to lush vegetation. Fredrickson and Reid (1986) reported that moist-soil impoundments in Missouri attracted over 150 avian species from 14 different orders.

Although naturalistic-management methods are normally preferred (i.e., water manipulation and natural-seedbank germination), mechanical and chemical methods may be required to manipulate successional stage (Fredrickson and Taylor 1982) and/or control undesirable species such as willow and cottonwood seedlings (Fredrickson and Reid 1988b), phragmites (Cross and Fleming 1989), and purple loosestrife (Thompson 1989). An expert computer model has been developed to help manage complexes of moist-soil impoundments (Auble et al. 1988), and computer software is available for estimating seed production of moist-soil plants (Laubhan 1992). Despite these technological advances, moist-soil management continues to be more of an art than a science (Fredrickson and Reid 1986) and must be tempered by a manager's experience in a specific locale (Reid et al. 1989).

Greentree reservoirs

Greentree reservoirs (GTRs) are intensively managed bottomland-hardwood wetlands. Levees and water-control structures are used to enhance the reliability of water and food supplies for migrating and wintering waterfowl (Mitchell and Newling 1986; Reinecke et al. 1989; Weller 1990), although a multitude of other species also benefit (Fredrickson and Batema 1992). In some areas (e.g., where bottomland hardwoods have been extensively drained and converted to other uses), GTRs provide the only habitat consistently available to migrating and wintering wildlife (Fredrickson and Batema 1992). Some of these areas are further enhanced by combining moist-soil management, in created openings, with traditional GTR management that emphasizes mast production (Harrison and Chabreck 1988; Weller 1990). This diversity of food resources and habitats is especially important to migrating- and wintering-waterfowl species that exhibit seasonal changes in diet and patterns of habitat use (e.g., Figure 12).

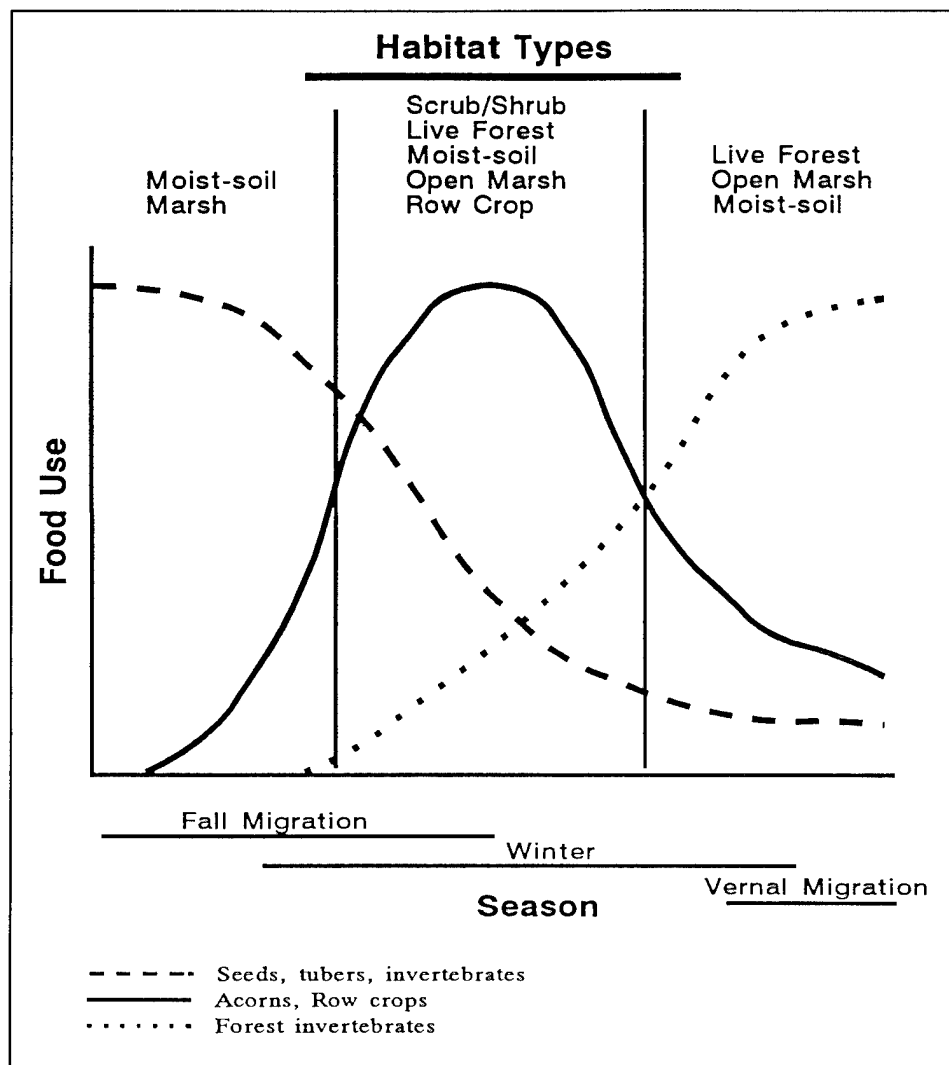


Figure 12. Mallard food use and habitat preferences during winter in upper portions of the Mississippi Alluvial Valley (from Fredrickson and Batema (1992:24))

Site selection for impoundments is crucial and must include mast-producing species that tolerate prolonged flooding (e.g., willow and water oak) but produce fruit or seed of a size suitable for ducks (Allen 1980; Weller 1990). Traditionally, GTRs were flooded annually after tree growth had declined in autumn and drained before growth resumed in spring (Reinecke et al. 1989). Although annual flooding provides important benefits, it also has potential long-term impacts and problems (Table 17). It is now recommended that GTRs be flooded once every 2 to 3 years instead of annually (Figure 13). This allows impoundments to dry out, which simulates natural conditions and, possibly, helps decomposition and nutrient cycling (Reinecke et al. 1989; Fredrickson and Batema 1992). Moreover, this practice encourages seedling establishment (Payne and Copes 1986; Payne 1992:232) and helps prevent the forest community from converting to vegetation characteristic of wetter habitats (Mitchell and Newling 1986).

Table 17
Benefits and Problems Associated with Greentree Reservoir
Management and Operation¹

| Benefits | Problems |
|---|---|
| Initial increase in use by waterfowl | Decline in waterfowl use over time |
| Increase in viable acorn production | Decline in acorn production for some oak species |
| Initial increase in radial growth for some tree species | Decrease in radial growth for some tree species |
| Increase in fire protection | Lack of regeneration of desirable mast species |
| Control of timing and duration of flooding | Burrowing animals compromise levees; beavers alter flooding regimes |
| Consistent supply of food and cover in fall | Flooding stress, disease, morphological changes and tree mortality |
| Acorns available sooner and longer; invertebrates larger in size, available sooner and longer | Lower plant species diversity and plant community changes to more water-tolerant forms; possibly lower invertebrate species diversity |
| ¹ Source: Fredrickson and Batema (1992:3). | |

Management Techniques

Techniques for wetland management include (a) developing areas for manipulating water levels, (b) establishing water on areas without the capability of manipulating water levels, (c) establishing and/or controlling vegetation, (d) controlling wildlife, and (e) providing artificial nesting and loafing sites (Payne 1992:4). The development of areas for manipulating water levels usually involves the construction, installation, and maintenance of dams, spillways, dikes, levees, and water-control structures (e.g., radial gates, roller gates, sliding gates, stoplog structures, overflow tubes, drop inlets, tin whistles, and pumps). Design criteria depend on management objectives, impoundment size, site characteristics, and water availability and reliability. Construction techniques have been described elsewhere (Linde 1969; Atlantic Waterfowl Council 1972; Payne 1992) and are not reviewed here. Instead, methods are reviewed for managing water levels and vegetative communities, which, in turn, influence abundance and species richness in the animal community.

Water-level manipulation

Water-level manipulation is the most frequently used technique to manage wetland-plant communities (Payne 1992:175), at least in wetlands with a regular supply of water and suitable water-control structures (Weller 1982). The main objective in water-level regulation is to maximize food resources and structural diversity. There are two general strategies: (a) maintain constant water levels or (b) fluctuate water levels through periodic drawdown and flooding events

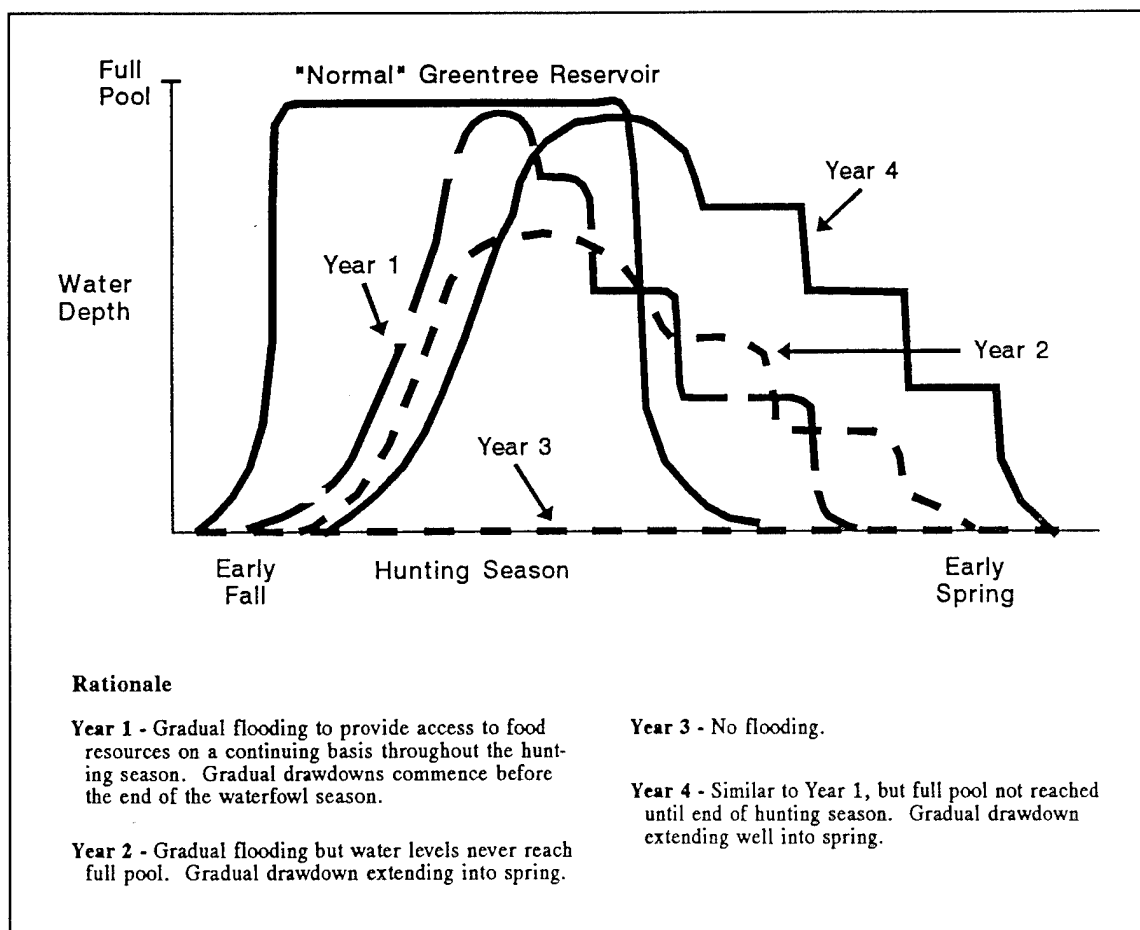


Figure 13. Flooding regimes suggested for southern greentree reservoirs (from Fredrickson and Reid (1988c))

(Atlantic Waterfowl Council 1972; Payne 1992:176). The appropriate strategy for a particular wetland will depend on management objectives, water and soil characteristics, and reliability of the water supply. For example, constant water-level management can be used to promote growth of submergent plants such as sago pondweed, which attract diving ducks and provide food and structure for the invertebrate community (Weller 1987). However, stabilizing water at high levels may eventually lead to lake-like conditions where production is aquatic rather than semiaquatic (Weller 1987:70). Furthermore, many freshwater marshes depend on annual and seasonal water-level fluctuations to maintain long-term productivity, although this should not be interpreted as a need for erratic water-level manipulation at any time of the year.

Water-level manipulation can be used to encourage or discourage vegetation. For example, flooding may be necessary to discourage dense vegetation or undesirable species that became established during dry periods. Conversely, drawdowns may be necessary for marsh revegetation, i.e., drawdowns promote persistent- and nonpersistent-emergent vegetation used for food and cover (e.g., nesting sites and brood cover). Drawdowns may be either complete or partial. Complete drawdowns are used when major restoration is needed in an open

marsh, i.e., when all current vegetation has been lost because of herbivory (e.g., muskrat eat-outs), high-water levels, large carp populations, winter kill, or, occasionally, plant diseases. Partial drawdowns are normally used to promote vegetation and discourage herbivores, i.e., when vegetation is seriously reduced, wildlife use has declined, or water levels have stressed vegetation. Weller (1982:950, 1987:75-77) described general guidelines for complete and partial drawdowns in freshwater marshes managed for breeding wildlife:

Complete drawdowns

- a. Lowering water levels permits the germination of naturally occurring seed and the recovery of established but flood-stressed emergents and submergents. Although collecting, growing, and planting seeds and tubers once was common, natural supplies usually are adequate.
- b. The degree of drawdown depends on the basin shape and water availability, but decomposition of bottom vegetation and cracking of bottom mud is ideal [conditions] for most plants.
- c. The length of drawdown is uncertain, but drying of the soil and breakdown of vegetation to release bound nutrients may require most of the growing season. Over-winter drawdowns often have proved effective. In certain settings, late fall or early winter (post-hunting) drawdowns can be left until reflooding in late summer (pre-hunting), so that duck hunting is not seriously affected. Muskrats will be drastically reduced over winter, however, which may be desirable for marsh management but less popular for muskrat trappers. Timing of the drawdown may be used to make trapping more successful and satisfy both user groups.
- d. Reflooding should be a gradual process, to avoid flotation of emergents, direct scouring of other plants, or plant mortality due to the turbidity of muddy waters. Late-summer flooding may induce muskrat use if depths are freeze-proof in northern latitudes. Keeping water levels low may attract birds but will not attract muskrats.
- e. Water levels should be regulated mainly for vegetation growth, diversity, and survival during the second (first reflooded) season, as long-term trends demonstrate a gradual decline of emergent vegetation with stable or high water levels. Some concern for wildlife must be deferred at this time, since long-term production of wildlife will be enhanced in later seasons.
- f. Subsequently, marsh management for the benefit of wildlife will consider welfare of the vegetation, but manipulation will be designed mainly to regulate muskrat use and enhance bird use. Knowledge of species requirements is essential, but the "system" will be self-forming and dynamic.
- g. Some submergents may germinate on mud flats, but most germinate underwater because they are better adapted to aquatic conditions.

Excessive depths, however, especially of turbid water, are detrimental to submergents.

- h.* After several years, reduction of muskrats or carp via late fall drawdown may conserve vegetation. Not all muskrat populations “explode,” but many do, and lowered water level increases their vulnerability to traps. Carp may freeze out over winter if depths are kept shallow.
- i.* Some small fish live in marshes but the consequences of drawdowns do not seem to have been documented. Marshes associated with lakes often are major spawning beds [for game species such as northern pike] and are, therefore, of importance in planning marsh drawdowns.

Partial drawdowns

- a.* Water levels should be reduced to meadowlike depths to encourage propagation of emergents and germination and growth of submergents in early summer, especially at the marsh perimeter. Wildlife use by species such as inland diving ducks and coots that favor deep water will decline; use by waders, shorebirds, and dabbling ducks may increase markedly.
- b.* This low water level should be retained or even lowered in late summer, and returned to near-normal levels in early fall.
- c.* It is best not to keep freeze-proof depths, except where plant density is high and muskrats are to be encouraged. The presence of carp is a consideration.
- d.* As vegetation recovers, levels are regulated to allow nesting or plant consumption as desired.

Similar or modified versions of these guidelines have been described by Linde (1969), Yoakum et al. (1980), and Payne (1992). Water-level management has also been described for specific-wetland systems: (a) moist-soil impoundments (Fredrickson and Taylor 1982); (b) GTRs (Fredrickson and Batema 1992); (c) tidal marshes (e.g., Atlantic Waterfowl Council 1972; Chabreck, Joanen, and Paulus 1989); and (d) other types of wetlands managed for wintering and migratory birds (Smith, Pederson, and Kaminski 1989). However, marshes vary considerably in response to water-level changes, even within the same State or region (Payne 1992). Consequently, successful water-level management requires a detailed understanding of the physical and biological characteristics of each marsh. For example, factors such as bottom topography, soil characteristics, existing plant communities, current waterfowl use and productivity, and water supply and flow rates all affect how and if water regulation is used as a habitat-manipulation technique (Yoakum et al. 1980). Readers are urged to consult Linde (1969), Atlantic Waterfowl Council (1972), Weller (1982, 1987), Smith, Pederson, and Kaminski (1989), and Payne (1992) for more detailed information.

Prescribed burning

Fire is an important tool in habitat management, but it must be carefully controlled and applied. Ward (1968) reported that burning dense stands of phragmites in large permanent marshes can enhance nutrient cycling and stimulate growth of vegetation attractive to nesting ducks. Furthermore, prescribed burns can (a) make new green shoots, roots, and rhizomes of grasses and sedges available to geese; (b) expose fallen seed for ducks; (c) eliminate sour-marsh conditions caused by flooded and decomposed organic matter; (d) reduce impenetrable growth of climax-plant species such as phragmites, bulrush, sawgrass, cordgrass, and cattail; (e) promote growth of seed-producing plants; and (f) create open pools and edge for nesting and feeding waterfowl (Payne 1992:232-233; Wright and Bailey 1982; Gordon et al. 1989). However, careless burning during the nesting season can result in direct wildlife losses (Cartwright 1942; Hochbaum, Kummen, and Caswell 1985), reduce snow-trapping ability in seasonal wetlands (Kantrud 1986; Pederson, Jorde, and Simpson 1989), and reduce plant-species diversity in nutrient-poor wetlands such as bogs (Mallik and Wein 1986; Payne 1992:233). Linde (1969) and Payne (1992) provided a more detailed review of the use of fire in marsh management (including adjacent uplands), and Kirby, Lewis, and Sexson (1988) compiled an extensive bibliography on fire and fire-wildlife relations in North American wetland ecosystems.

Control of herbivores

Controlled grazing can be used to open up dense patches of cover (Kantrud 1986; Rutherford and Snyder 1983; Payne 1992) and promote growth of annuals while reducing growth of undesirable species such as willow (Chabreck, Joanen, and Paulus 1989). However, many wetland-plant species respond differently to grazing, mowing, burning, and tillage (Payne 1992:234-235). The type of grazer (e.g., sheep, horses, and cattle) also can make a difference. Horses control woody vegetation better than cattle (Pederson, Jorde, and Simpson 1989), but sheep can be more easily managed (Ermacoff 1968). However, intense sheep grazing may reduce diversity of nesting birds (Daiber 1986).

Regardless of the species, grazing intensity and duration must be carefully controlled, especially during drought. For example, Weller, Wingfield, and Low (1958) reported a negative impact of cattle grazing on duck populations during a severe drought, and Kirsch (1969) described negative effects of grazing on ducks that nest in upland vegetation. Similarly, Weller (1982) reported that cattle can have a great impact by creating trails and paths through wetland vegetation.

Payne (1992) recommended maintaining a mosaic of habitats in areas managed primarily for nesting birds. He suggested that such a mosaic could be produced by combinations of varying intensity and frequency of grazing and trampling, mowing, and burning. Wild herbivores such as geese, beaver, muskrats, and nutria can also be used as a management tool; however, population control can be very difficult (Weller 1987; Payne 1992).

Control of exotic and other nuisance species

The introduction or invasion of nonnative species such as carp and purple loosestrife can have widespread, detrimental effects on wetlands and waterfowl habitat (Ratti and Kadlec 1992). Control methods for nuisance species have been developed, but they often are expensive and temporary. Carp-control techniques include winter drawdowns, rotenone poisoning, and elimination of reinvasion routes (Kadlec and Smith 1992). Control of purple loosestrife and other noxious plants includes use of herbicides (e.g., Rodeo), mowing and tillage, planting Japanese millet on drawdown sites, and avoiding perturbations to native vegetation that would allow the plant to become established (Thompson 1989). Experiments are also being conducted with biological-control agents (i.e., introduction of host-specific organisms that feed on the noxious-plant species). Although this is the preferred ecological approach, many nuisance species require integrated control (i.e., combinations of mechanical, chemical, and biological methods). There are several reports in the USFWS's Waterfowl Management Handbook (Cross 1988) that discuss control methods for phragmites, cattail, and purple loosestrife.

Mechanical treatment of vegetation and substrate

Mechanical methods can be used to simulate the open-water phase in larger marshes that lack water-control structures (Weller 1982); however, the design will depend on management objectives. For example, Kaminski and Prince (1981) recommended that artificially created openings in wetlands managed for breeding waterfowl should be (a) randomly spaced circles at least 0.1 ha to reduce aggregations of breeding ducks and allow diving ducks to take flight, or (b) shaped in sinuous strips to increase edge and reduce visual encounters between conspecific pairs of ducks. Openings can be created through cutting or mowing, herbicides, basin deepening, or a combination of methods (Linde 1969; Beule 1979; Payne 1992).

The most common approach is basin deepening, which usually involves dredges, draglines, bulldozers, or blasting (Weller 1987:78). One frequently used method in near-dry marshes >0.8 ha is level ditching. Level ditches usually are dug with a dragline, although marsh-buggy plows, marsh cutters, backhoes, and blasting also have been used (Atlantic Waterfowl Council 1972; Martin and Marcy 1989; Payne 1992:130). Level ditching has shown positive results for muskrat and waterfowl populations; however, it can be aesthetically unpleasing (Weller 1987:79). Other methods such as blasting (with dynamite and/or ammonium nitrate) and bulldozing may be less expensive and more aesthetically satisfying, but they generally modify too little of the marsh bottom to create cover-water interspersions comparable to a natural hemimarsh (Weller 1982). Payne (1992) provided a detailed review of the various methods for modifying basin structure.

Openings may also be created via mechanical and chemical treatment of the vegetation. For example, openings can be created in dense stands of cattail by burning in the winter, shredding remaining plant stalks with a rotary mower, and

then flooding the cut stalks for at least 2 weeks in the spring (Payne 1992:258; also see Weller 1975; Murkin and Ward 1980). Other methods include (a) cutting cattail during winter and then flooding over the stubble in spring (Linde 1969; Weller 1975; Buele 1979); (b) March drawdowns combined with burning, plowing or discing, and reflooding (Hindman and Stotts 1989); and (c) summer cuttings when carbohydrate reserves are low (Linde, Janisch, and Smith 1976; Payne 1992:258). Short-term programs also have been developed to control phragmites, rice cutgrass, and undesirable species such as willow, cocklebur, and buttonbush.

Mechanical methods have also been used in smaller marshes to create and maintain diverse complexes of wetland habitats in the landscape. For example, small ponds for pairing ducks can be created by blasting or bulldozing in shallow-marsh habitats such as sedge meadows. However, this may not always be advisable because (a) shallow-marsh habitats support nontarget species such as the sedge wren and (b) during high-water levels these marshes may change from shallow to deep-marsh habitats and attract species that have shifted from marshes that are now less attractive (because of high-water levels or muskrats) (Weller 1982).

Chemical treatment of vegetation

Plant control with herbicides is relatively simple and inexpensive (Payne 1992:272); however, herbicides must be used with extreme caution because they are difficult to apply effectively (e.g., herbicide/pesticide drift is a common problem). Furthermore, many chemicals are nonselective; that is, they often kill more than target species (Weller 1987:80; Payne 1992:271). In some cases, however, herbicides may be the only way to control undesirable species, especially nonnative plants such as water hyacinth and purple loosestrife. Herbicides are used alone, or in combination with mechanical treatments and biological controls, to accomplish the following objectives (Rollings and Warden 1964; Payne 1992:271-272):

- a. Create open-water areas in dense, emergent vegetation for general use by all waterfowl and nesting coots and diving ducks.
- b. Create open-loafing areas in shoreline vegetation and on nesting islands and inaccessible sandbars.
- c. Destroy emergent vegetation used as predator travel lanes to reach nesting islands.
- d. Reduce nesting cover on dikes to reduce duck nesting there and consequent nest losses to predators using the dikes as travel lanes.
- e. Facilitate maintenance of dikes, ditches, canals, and water control structures by destroying woody vegetation.

- f. Control algae to reduce potential algae poisoning and to improve light penetration and growth of food plants, which also provide cover for invertebrates.

Relatively few herbicides are registered for use in or near aquatic areas. These can be subdivided into (a) herbicides approved for aquatic use in water supporting food fish (e.g., copper sulfate, diaquat, endothal, 2,4-D, and Rodeo) and (b) herbicides approved for aquatic use in water without food fish and having other possible restrictions (e.g., bromacil) (Schnick et al. 1982; also see USFWS 1979; Hansen, Oliver, and Otto 1984). Payne (1992) reviewed herbicides and their application in wetland management.

Artificial nesting structures

A common practice in wetland management is the establishment of artificial nesting structures such as man-made islands (Linde 1969; Yoakum et al. 1980). Nesting islands can provide security from predators and increase suitable-nesting area in open-water locations (Payne 1992:339); however, an island's effectiveness depends upon size, location, maintenance, local-predator populations, and behavior of species using the island (e.g., some species are more aggressive than others). Payne (1992) reviewed the creation, placement, and maintenance of nesting islands, including earthen islands, rock islands, culverts, bales, brush islands, and floating islands.

A similar practice in wetlands is to provide nest boxes for wood ducks and hooded mergansers, nesting platforms and baskets for Canada geese and mallards, and loafing logs for ducks and turtles. However, these structures can be expensive to build and maintain. Furthermore, installing artificial nesting structures does not always result in increased production by the target species (e.g., Soulliere 1986). On the other hand, placement of nesting structures in areas where natural nesting sites are absent or lacking, or where predation rates are high, can be both cost productive and ecologically beneficial (e.g., Mackey, Mathews, and Ball 1988). Moreover, building and installing nesting structures can be a means to promote conservation interests among local conservation groups, land owners, and the general public.

Determining and estimating the availability of natural sites relative to space and other resource requirements of the target wildlife is recommended before installing nesting structures. Payne (1992) reviewed the construction and placement of artificial structures, including designs for nongame species such as ospreys, colonial waterbirds, cliff swallows, common loons, and Everglade kites. Linde (1969) and Yoakum et al. (1980) also reviewed nesting structures and man-made islands.

Limits of Management

Wetland management is based on knowledge of plant succession, wildlife-habitat requirements, and interactions among wildlife, vegetation, and wetland

dynamics. However, management outcomes are not predictable to a high degree of accuracy because all environmental influences are not understood and cannot be controlled (Weller 1987:85). Moreover, much knowledge has been derived from observational studies or simply trial-and-error. Experimental evidence from large-scale manipulations with regulated controls has been lacking. One exception is the recently completed Marsh Ecology Research Program (MERP) (see Murkin and van der Valk 1984; Weller 1987). A series of man-made cells were constructed in the Delta Marsh, Manitoba, Canada. These cells (i.e., marsh-management units) permitted replicated studies of wetland manipulations such as water level, mowing, and fertilization. Ecological responses (i.e., changes in primary production, nutrient cycling, vegetative characteristics, avian composition, invertebrate density and diversity, etc.) were scientifically monitored and compared among the various treatments. "Only with such data can precise modeling and prediction become reality" (Weller 1987:87).

However, more information is needed, especially in different wetland systems. Weller (1982:953) described several topics that need careful experimental study:

- a. Habitat stimuli that attract wildlife to marshes.
- b. The development of indices to wildlife production in marshes.
- c. The size of isolated areas essential to the development and/or maintenance of marsh fauna (i.e., the marsh as a habitat "island").
- d. The diversity or heterogeneity of wetland areas in a complex essential to attract and maintain marsh wildlife.
- e. Wetland:upland ratios conducive to preservation of typical prairie-wetland biotas.
- f. Germination conditions that make marsh drawdowns or other water manipulations more effective and predictable.
- g. A better understanding of water and soil chemistry of marsh systems.
- h. The role of siltation, fertilizers and other man-made products in modifying productivity of wetland areas.
- i. Objective experimentation on grazing, burning, and other natural procedures to assess the role in marsh management for wildlife.
- j. The relationship of invertebrates to marsh dynamics.
- k. Detailed studies of the biology of dominant aquatic plants, such as the work of Linde, Janisch, and Smith (1976).

Studies such as MERP have provided important information on some of these topics; however, many data gaps remain. An adaptive-management strategy

combined with careful experimental studies is needed to advance wetland-management skills and make management a more predictable science. As Weller (1987:87) suggested, "each management operation should be part of long-term program with preliminary observations, records of actions taken, and a follow-up measuring success—regardless of how superficial they may seem."

Despite these limitations, the novice can achieve fair success at managing a small-marsh area to improve cover diversity and enhance wildlife populations (Weller 1987). To be successful, small-scale programs require (a) an assessment of what is desirable and good for wildlife (i.e., sound objectives must be established before implementing a management program); (b) some observations of natural succession and other wildlife processes, either under different conditions or with several types of wetlands as examples; (c) some logic in assessing important environmental influences such as water depth; and (d) some modest experimentation (Weller 1987:87). In contrast, large-scale-management programs (i.e., large marshes or complexes of wetlands) often must deal with more complicated issues such as conflicts among management-oriented interest groups (e.g., homeowners, boaters, fisherpersons, birdwatchers, and hunters) and limited financial resources.

There are no rigid guidelines for managing wetland ecosystems. Hence, managers must be creative, adopt a flexible-management strategy, and make onsite decisions based on their own expertise. In other words, "good habitat management requires a manager who recognizes the seasonal needs of the birds [and other vertebrate and invertebrate species], knows the ecology of the local marsh ecosystems and adjacent uplands, and then applies appropriate principles to develop methods suitable for the local situation" (Kadlec and Smith 1992:590). The expertise needed to make effective management decisions takes years to develop and represents a combination of continuing education and field experience (Laubhan and Fredrickson 1993). Books, professional papers, and management manuals offer valuable guidance; however, they are no substitute for getting into a wetland and learning the ecology of a particular system. Furthermore, management is an ongoing process in which life-history characteristics and habitat requirements of many species are unknown. Consequently, managers must be prepared to incorporate information on a species' biology and habitat requirements into their management plans when it becomes available.

8 Wetland Conservation and Protection

Prior to the 1970s, there was little consideration given to the impacts of development on wetlands and their associated flora and fauna. The Fish and Wildlife Coordination Act (16 U.S.C.A. 611 et seq.), first passed in 1934 and amended several times after World War II, was one of the first pieces of legislation to require Federal water-development agencies (e.g., USACE) to consider, where feasible, wildlife protection and mitigation measures in water-project plans. Although this act was a step in the right direction, conservation measures were often overlooked or only partly implemented when the water projects were constructed (Smythe 1989:10). Congress did not pass stronger environmental legislation until 1969.

In 1969, Congress passed the National Environmental Policy Act (NEPA) (42 U.S.C.A. 4321 et seq.), which established procedures for evaluating alternatives and developing mitigation plans for environmental impacts associated with Federal projects. More importantly for wetland conservation, the statute opened the USACE's planning process to public review and comment and required that attention be given to the environmental effects of proposed actions such as dredge and fill, hydroelectric, and flood-control projects (Smythe 1989:39). This significantly affected the USACE's planning process for proposed water projects.

Congress passed another significant piece of environmental legislation in 1972. The Federal Water Pollution Control Act Amendments of 1972 (33 U.S.C. 1251 et seq.), commonly called the Clean Water Act, transferred authority to regulate most point-source pollutant discharges into the "waters of the United States" from the USACE to the USEPA. However, Section 404 of the act established a permit program through which the USACE was to regulate the discharge of dredged and fill material into United States waters. Although Section 404 was initially interpreted to apply only to navigable waters, court decisions have established that most streams, lakes, and wetlands are also subject to Section 404 (Smythe 1989:11).

In May 1977, President Jimmy Carter issued two executive orders that established the protection of wetlands as an official policy of the Federal government. These orders were significant because they caused Federal agencies to review their wetland and floodplain policies (Mitsch and Gosselink 1986:443). Today, there are a number of Federal directives, statutes, and programs that

directly or indirectly protect wetlands in the United States. There also is a diverse mixture of State and private programs designed to preserve and enhance wetland resources. However, there is no specific national wetland law. As Mitsch and Gosselink (1993:565) concluded:

Wetland management and protection result from the application of many laws intended for other purposes. Jurisdiction over wetlands has also been spread over several agencies, and, overall, federal policy continually changes and requires considerable interagency coordination.

In the following sections, some regulatory programs, acquisition and incentive programs, and programs that promote the conservation, protection, and, in some cases, restoration of wetlands are reviewed. More detailed reviews can be found in the following sources. Mitsch and Gosselink (1986, 1993) reviewed national policies and laws affecting the legal protection of wetlands in the United States. The OTA (1984) described Federal programs and some State, local, and private initiatives for wetland acquisition and protection. Salvensen (1990) discussed the regulation and mitigation of developmental impacts. Ratti and Kadlec (1992) reviewed acquisition, easement, and enhancement programs, with special emphasis on programs affecting wetlands in the Intermountain West. The NRC (1992a) reviewed Federal programs for wetland restoration. Finally, several papers in Hook et al. (1988b) discussed protection and management of wetland resources in the United States.

Regulatory Programs

Section 404 of the Federal Water Pollution Control Act (1972) and the 1977 Amendments are the Federal government's primary tool for regulating the discharge of dredged or fill material in wetlands. The 404 Program is administered by the USACE with assistance from the USEPA. The 404 regulations prohibit discharge of dredge-and-fill materials into wetlands without a permit from the USACE. Other programs (e.g., Swampbuster) address excavation, drainage, clearing, and flooding of wetlands not covered explicitly under the 404 Program.

The Swampbuster provision of the 1985 Food and Securities Act (as amended in the 1990 Farm Bill) specifies that any person who converts wetlands to agricultural use (i.e., commodity production) after December 23, 1985, becomes ineligible for most Federal agricultural subsidies (Ratti and Kadlec 1992). The Swampbuster program, along with changes in the tax treatment of agricultural drainage (Tax Reform Act of 1986), amendments to the Clean Water Act (Section 404 Program), and lower grain prices have substantially reduced wetland-conversion rates since the early 1980s (NRC 1992a:285). However, jurisdictional protection of these smaller and less permanent wetlands depends on exact definitions and delineation criteria. For example, recently proposed changes to the *Federal Manual for Identifying and Delineating Jurisdictional Wetlands* could result in 50 to 80 percent of the nation's wetlands losing jurisdictional protection (Committee on Science, Space, and Technology 1991).

Vulnerable wetlands would include seasonal and temporary ponds, vernal pools, meso- or xero-riparian habitat, and portions of the Florida Everglades and Virginia's Great Dismal Swamp. Until this issue is resolved, agencies involved have agreed to use the 1987 Wetlands Identification and Delineation Manual (Williamson 1993).

A number of other Federal laws, directives, regulations, and programs also affect wetland management and protection (Table 18). Some of these affect wetlands directly (e.g., Coastal Zone Management Act of 1972), whereas other programs protect wetlands indirectly through water quality standards (e.g., Sections 208, 303, and 402 of the Clean Water Act of 1977), land-conservation efforts (e.g., Conservation Reserve Program), and preservation of habitat for endangered species (Endangered Species Act of 1973, as amended). In addition, many States have developed comprehensive wetland laws for inland waters (Table 19).

Acquisition and Incentive Programs

Several Federal and State programs protect wetlands through acquisition (ownership, lease, or easement) or incentives. The more well-known programs include the establishment of the Migratory Bird Hunting and Conservation Stamp, the Land and Water Conservation Fund Act, the Water Bank Program, and the Conservation Reserve Program.

Migratory bird hunting and conservation stamps (1934)

Proceeds from the sale of "duck stamps," which must be purchased by waterfowl hunters aged 16 years or older, are used to acquire habitat for migratory birds. By 1984, over 1.4 million ha of waterfowl habitat had been preserved (Mitsch and Gosselink 1986:7), a large portion of which is wetland (OTA 1984:72). Duck stamp receipts also are used to repay appropriations from the Wetlands Loan Act (1961), which provided interest-free-loan advances for wetland acquisition and easements (OTA 1984).

The Land and Water Conservation Fund Act (1965)

This program funded the purchase of many natural areas, including wetlands. The USFWS used this source of funding to protect endangered species and important natural areas and to extend the National Wildlife Refuge System (OTA 1984).

Table 18
Major Federal Laws, Directives, Regulations, and Programs Used for the
Management, Protection, and Restoration of Wetlands¹

| Directive, Statute, or Program | Date | Responsible Agency |
|---|------------|---|
| Rivers and Harbors Act | 1899 | Army Corps of Engineers |
| Fish and Wildlife Coordination Act | 1934, 1967 | U.S. Fish and Wildlife Service |
| Water Resources Planning Act | 1965 | Departments of Agriculture, Interior, Army, and Health, Education, and Welfare |
| Land and Water Conservation Fund Act | 1968 | U.S. Fish and Wildlife Service, Bureau of Land Management, National Park Service |
| Water Bank Program | 1970 | Agriculture Stabilization and Conservation Service, with assistance from the Soil Conservation Service and the U.S. Fish and Wildlife Service |
| Federal Water Pollution Control Act and Amendments | 1972, 1977 | |
| Section 404 - Dredge and Fill Permit Program | | Army Corps of Engineers with assistance from the Environmental Protection Agency |
| Section 208 - Areawide Water Quality Planning | | Environmental Protection Agency |
| Section 303 - Water Quality Standards | | Environmental Protection Agency |
| Section 401 - Water Quality Certification | | Environmental Protection Agency (with State agencies) |
| Section 402 - National Pollutant Discharge Elimination System | | Environmental Protection Agency (with State agencies) |
| Coastal Zone Management Act | 1972 | Office of Coastal Zone Management |
| Flood Disaster Protection Act | 1973, 1977 | Federal Emergency Management Agency |
| Endangered Species Act | 1973 | U.S. Fish and Wildlife Service, National Oceanic and Atmospheric Association |
| Federal Aid to Wildlife Restoration Act | 1974 | U.S. Fish and Wildlife Service |
| Water Resources Development Act | 1976, 1990 | Army Corps of Engineers |
| Executive Order 11990 Protection of Wetlands | May 1977 | All Federal agencies |
| Executive Order 11988 Floodplain Management | May 1977 | All Federal agencies |
| Food Securities Act, Swampbuster Provision | 1985 | U.S. Department of Agriculture, Soil Conservation Service |

(Continued)

¹ Source: after Mitsch and Gosselink (1993:566), with additional data from OTA (1984), Smythe (1989), NRC (1992a), and Ratti and Kadlec (1992).

| Table 18 (Concluded) | | |
|---|--------------------|---|
| Directive, Statute, or Program | Date | Responsible Agency |
| Emergency Wetland Restoration Act | 1986 | U.S. Fish and Wildlife Service |
| North American Waterfowl Management Plan | 1986 | U.S. Fish and Wildlife Service, Canadian Wildlife Service |
| Wetland Delineation Manuals (and various revisions) | 1987 1989, 1991 | Army Corps of Engineers, Environmental Protection Agency, Soil Conservation Service |
| North American Wetlands Conservation Act | 1989 | U.S. Fish and Wildlife Service |
| "No Net Loss" Wetlands Policy | 1988 | All Federal agencies |
| Coastal Planning, Protection, and Restoration Act | 1990 | Army Corps of Engineers, Environmental Protection Agency |
| Wetland Reserve Program | 1991 | U.S. Department of Agriculture, Soil Conservation Service |
| Office of Wetland Protection | | Environmental Protection Agency |
| Partners for Wildlife | | U.S. Fish and Wildlife Service |
| Taking Wing | | U.S. Forest Service |

Table 19
States That Have Comprehensive Wetland Laws for Inland Waters¹

| State | Law |
|---------------|--|
| Connecticut | Inland Wetlands and Watercourses Act |
| Delaware | The Wetlands Act |
| Florida | Henderson Wetlands Protection Act of 1984 |
| Maine | Protection of Natural Resources Act |
| Maryland | Chesapeake Bay Critical Area Act |
| Massachusetts | Wetland Protection Act |
| Michigan | Goemaere-Anderson Wetland Protection Act |
| Minnesota | The Wetland Conservation Act of 1991 |
| New Hampshire | Fill and Dredge in Wetlands Act |
| New Jersey | Freshwater Wetlands Protection Act of 1987 |
| New York | Freshwater Wetlands Act |
| North Dakota | No Net Wetlands Loss Bill of 1987 |
| Oregon | Fill and Removal Act Comprehensive Land Use Planning Coordination Act |
| Rhode Island | Freshwater Wetlands Act |
| Vermont | Water Resources Management Act |
| Wisconsin | Water Resources Development Act Shoreland Management Program |

¹ Source: Mitsch and Gosselink (1993:575); copyright 1993 by Van Nostrand Reinhold, reprinted with permission.

Water Bank Program (1970)

Objectives of this program are to preserve, restore, and improve wetlands of the Nation. The program is administered by the Agricultural Stabilization and Conservation Service (ASCS) with technical assistance from the U.S. Soil Conservation Service. It is a 10-year agreement between private-wetland owners (they need not be agricultural producers) and the U.S. Department of Agriculture (USDA) to carry out restoration and management practices that promote waterfowl production and other wildlife benefits (Ratti and Kadlec 1992). Landowners or operators normally receive annual payments in exchange for agreeing not to drain, fill, level, burn, or otherwise destroy wetlands and to maintain grassy cover on adjacent-upland areas. The program has been most successful in the prairie-pothole region of Minnesota, North Dakota, and South Dakota (OTA 1984).

Other Federal programs

The Partners for Wildlife program provides opportunities for preserving, restoring, creating, and enhancing wetland habitat on private lands. The USDA Wetland Reserve Program is a voluntary program in which eligible landowners (farmers) receive cash payment for restoring and protecting wetlands on their property. It is similar to the Conservation Reserve Program but focuses more on wetlands and requires longer term easements (i.e., ≥ 30 years). Lands eligible for the Wetland Reserve Program include farmed wetlands that are restorable and wetlands converted to cropland prior to December 23, 1985. Stream corridors (riparian areas) that connect protected wetlands also are eligible. The program is administered through the Agriculture Stabilization and Conservation Service.

The North American Wetlands Conservation Act (NAWCA) and the North American Waterfowl Management Plan (NAWMP) are two Federal programs that provide opportunities for protection, restoration, creation, and management of wetlands in the United States, Canada, and Mexico. The NAWCA provides matching grants to public-private partnerships for wetland projects that benefit waterfowl and other migratory birds. A nine-member council appointed by the Secretary of Interior recommends projects to the Migratory Bird Conservation Commission for approval of funding. Funding is administered by The North American Waterfowl and Wetlands Office of the U.S. Fish and Wildlife Service (Graziano and Cross 1993).

Projects proposed under the NAWCA must be consistent with the goals of the NAWMP, which is an ambitious wetland-waterfowl recovery plan to restore and maintain waterfowl populations and wetland habitats to a level common to the 1970s (USFWS and Canadian Wildlife Service (CWS) 1986). The key to achieving this goal is development of public-private partnerships (i.e., the joint-venture concept). Joint ventures are designed to maximize financial, organizational, and other in-kind support toward a common objective in a geographic region. Habitat joint ventures have been implemented in the following regions: Atlantic Coast, Central Valley of California, Eastern Provinces of Canada, Playa Lakes Region, Prairie Habitat Region of Canada, Prairie Pothole Region in the United States, Rainwater Basin of south-central Nebraska, and the Upper Mississippi River/Great Lakes Region (Graziano and Cross 1993).

State and private programs

A number of State programs (e.g., Minnesota Water Bank Program, California Permanent Wetland Easement Program, and Washington State Ecosystems Conservation Program) and private programs (e.g., Ducks Unlimited, The Nature Conservancy, and National Audubon Society) contribute directly and indirectly to wetland acquisition and protection. Many States also regulate wetland use through programs whose primary purpose is not wetlands protection, e.g., scenic and wild-rivers protection, critical or natural-areas protection, dredge-and-fill acts, wildlife and waterfowl protection,

stream-altercation requirements, and public-lands management (OTA 1984). Finally, programs such as the Ramsar Convention (Navid 1988) and the North American Waterfowl Management Plan (USFWS and CWS 1986) attempt to identify and protect critical-wetland habitat on a regional and international scale.

Restoration Programs

The United States has programs to restore and protect water quality (e.g., Clean Water Act) and to retard the loss of wetlands (e.g., 404-permit program and Swampbuster); however, few programs are designed specifically for wetland restoration. Furthermore, none of these programs promote large-scale, systematic-wetland restoration; such projects are left to a mixture of grassroots, local, and State initiatives (NRC 1992a:288-289). There are programs, however, that encourage small-scale, nonsystematic wetland restoration. These include the Section-404 program, Conservation Reserve Program, Wetlands Reserve Program, Water Bank Program, ASCS's WL-2 Shallow Water Areas practice (under the Agriculture Conservation Program), USFWS restoration projects (e.g., Partners for Wildlife), and a few USACE and Bureau of Reclamation projects.

There also are State programs and private organizations (e.g., Ducks Unlimited) that promote small-scale-wetland restoration. Such projects will likely increase in the future. For example, Ratti and Kadlec (1992) reported that Permit #27, which was added to Section 404 in November 1991, will promote restoration of altered and dredged nontidal wetlands on private, State, and Federal lands. Moreover, the NRC (1992a:286-287) suggested that Federal-water-development agencies (e.g., USACE, U.S. Bureau of Reclamation), in response to Congressional actions, will become more actively involved in wetland restoration.

The Nation's wetland policies and programs provide useful opportunities to restore small-wetland parcels; however, these programs are unlikely to restore large-wetland ecosystems that have been seriously degraded or to restore wetlands throughout a landscape (NRC 1992a:289). There currently is only one program (i.e., the Wetland Reserve Program) with the potential to promote large-scale, systematic restoration. Because the Wetland Reserve Program is directed at wetland systems and provides for conservation easements of 30 years or longer, it has the potential to promote large-scale restoration of aquatic ecosystems (NRC 1992a:288-289).

9 Summary

Freshwater wetlands are commonly referred to by such names as marshes, swamps, bogs, wet meadows, potholes, sloughs, and river-overflow lands. In general, wetlands can be defined as "those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions" (USEPA, 40 CFR 230.3, December 24, 1980; USACE, 33 CFR 328.3, November 13, 1986). In addition, most wetlands have substrate consisting of either predominately undrained hydric soil or a nonsoil that is saturated with water or covered by shallow water at some time during the growing season of each year.

Although this general definition covers most wetland types, one should be aware of ecologically important areas that may be excluded. For example, some floodplain habitats (i.e., riparian areas) may only be flooded for short periods during the nongrowing season of each year. These areas may not qualify as wetlands under current definitions; nevertheless, these "wet" areas support a wide diversity of plant and animal species and provide other valued functions such as groundwater recharge and floodwater storage. Moreover, these areas often have distinct properties (e.g., soil-moisture content, vegetative characteristics, diversity of plant and animal species) that differ from adjacent upland habitats.

There is no single, universally recognized definition that adequately describes all wetland types. The problem of definition arises because wetlands usually lie along a continuum between dry terrestrial ecosystems and permanently wet aquatic ecosystems. Consequently, wetlands are highly diverse in form and function. Furthermore, the reasons or needs for defining wetlands vary among interest groups. For example, wetland scientists need a flexible but rigorous definition that can be used in classification, inventory, and research; wetland managers are often concerned with regulations governing wetland modification and protection, and thus need clear, legally binding definitions; and policymakers need a definition that accommodates broad regional differences in wetlands and allows wetlands to be identified even in dry periods.

The problems of wetland definition also apply to wetland classification and inventory. For example, wetland classification involves artificially dividing up what is really an ecological continuum. Hence, some wetlands may seem to fit into more than one category and may be judged differently by different investigators. This problem is exacerbated by the dynamic nature of most wetlands. Depending on the classification criteria used, wetlands can at various stages, in even a few years of time, functionally span several categories. These changes are important because wetland types and habitats differ in their attractiveness to plant and animal species.

Wetland scientists have devised numerous classification schemes; however, most schemes focus on specific geographic areas or a restricted range of wetland types. A few wetland-classification schemes were designed for broad-scope coverage, but they have different goals, objectives, and classification criteria. Three broad-scope classification schemes that have influenced wetland management and regulatory decisions in the United States have been reviewed: (a) the early classification scheme used by the USFWS (Martin et al. 1953), (b) the classification scheme currently used by the USFWS (Cowardin et al. 1979), and (c) a hydrogeomorphic classification used by the USACE (Brinson 1993). Both USFWS classification schemes emphasized biotic characteristics of wetlands and were designed for national wetland inventories. The hydrogeomorphic classification emphasized abiotic features and was designed to support ongoing efforts to develop methods for assessing physical, chemical, and biological functions of wetlands.

Wetland classification is a necessary component in efforts to inventory our Nation's wetland resources. Inventories determine the extent of various types of wetlands in a given region. Inventories can be conducted at various levels of detail, depending on the specific needs of the user. Broad-scale inventories have been used to provide important information on status and trends of wetlands in the United States. For example, as part of the National Wetlands Inventory (NWI), the USFWS designed and implemented the first comprehensive, statistically valid effort to estimate the status of our Nation's wetlands. Results of this study indicated that inland, freshwater wetlands accounted for 95 percent of the estimated 41.8 million ha of wetlands in the conterminous United States in the mid-1980s. Of these, 52.9 percent were forested, 25.1 percent were emergent, 15.7 percent were scrub-shrub, and 6.3 percent were nonvegetated (e.g., open ponds and aquatic-bed areas). The NWI also indicated that wetland losses and alterations have been significant throughout the conterminous United States.

An estimated 53 percent of the original 89.4 million ha of wetlands in the lower 48 States were lost by the mid-1970s. In the most recent study, Dahl and Johnson (1991) reported a net loss of 1.1 million ha of wetlands from the mid-1970s to the mid-1980s. Freshwater wetlands accounted for 98 percent of this loss, with most (54 percent) losses resulting from conversion of wetlands to agricultural uses. However, wetland losses have not occurred evenly across the United States. From a regional perspective, the greatest rates of wetland loss occurred in the Lower Mississippi Alluvial Plain, the Pacific Mountains, the Gulf-Atlantic rolling plain, and the Gulf coastal flats. In absolute acreage, the

greatest losses of wetlands occurred in the Lower Mississippi Alluvial Plain, the Gulf-Atlantic rolling plain, and the Upper Midwest. Although legislation and conservation programs have been established to protect wetland ecosystems, wetland losses and alterations continue.

Human-induced impacts are responsible for most wetland alterations. These alterations can be grouped into three types: biological, chemical, and physical. Biological alterations frequently result from management that maximizes specific wetland values such as harvesting or removal of natural biota. Chemical alterations occur through point and nonpoint sources of pollution. Physical alterations include activities such as draining, dredging, and filling of wetlands. Physical alterations are often the most destructive because they frequently eliminate or significantly modify a wetland's hydrogeomorphology. However, biological, chemical, and physical alterations often occur together, and their collective impact may well be synergistic.

Wetland alterations are not always negative. For example, wetland management frequently involves the control or modification of water level, nutrient status, and natural disturbances, which are key factors affecting wetland ecology. Understanding the ecology of a wetland is a prerequisite to successful management, especially if the primary objective is to support a wide diversity of plant and animal species. One should be familiar with wetland concepts such as hydrogeomorphology, the role of wetland soils, wetland biogeochemistry, biological adaptations to wetland conditions, functional-biotic components, nutrient cycling, wetland dynamics (i.e., daily, seasonal, and longer term changes), and wetland functions. In addition, wetland managers and program administrators must be increasingly sensitive to the role wetlands play in biodiversity conservation.

Biological diversity (commonly termed biodiversity) is the variety of life and its processes. The concept involves multiple levels of organization (i.e., genetic, species, ecosystem, and landscape) and includes structural, functional, and compositional components. Ecological and evolutionary processes are also an important part of biodiversity. To understand biodiversity and its implications for land management, one must be aware of these factors and how they interact to affect communities and individual species. Moreover, this complexity should be considered when developing strategies to inventory, monitor, and assess biodiversity.

Wetland conservation and management should be an important part of efforts to maintain and protect biodiversity. Although wetlands only occupy about 5 percent of the land surface in the conterminous United States, they provide critical habitat for over 900 species of wildlife, including greater than one-third of the Federally listed endangered and threatened plants and animals. Riverine habitats and palustrine wetlands are especially vital to endangered and threatened species. Wetlands also perform other functions that indirectly support biodiversity conservation (e.g., surface-water storage, groundwater recharge, nutrient transformation and cycling, and maintenance of ecosystem integrity).

Several attributes of wetlands are particularly important for maintaining biodiversity: persistence of habitat, resiliency of the system, resistance to invasive species, nutrient cycling, productivity, ratio of emergent vegetation to open water, and ratio of wetland to upland habitat. Wetland juxtaposition and interactions with other ecosystems in the landscape are also important considerations. For example, the presence, location, and structure of wetland-habitat corridors may be especially important in heavily fragmented landscapes. For wetland-associated species, the importance of habitat complexes cannot be overemphasized. The protection and maintenance of different wetland habitats, or wetland types, arranged in close juxtaposition helps to maintain faunal and floral diversity typical of a region and avoids endangering some species while overproducing others. This diversity is important because a single-wetland type does not usually provide all resources required by different species, nor does a single-wetland usually provide the resources needed by various stages in the life cycle of one species. Consequently, the pattern and composition of a wetland complex can strongly influence species richness.

The greatest diversity of organisms is usually found in large wetland complexes with (a) a mixture of habitats ranging from open water and mudflats to dense rank vegetation, (b) a good interspersed of open water (50 to 70 percent) and emergent cover, and (c) relatively shallow water-levels (<45 cm). Individual sites may vary, however. For example, ideal water depths for a given area will depend on primary-wildlife users and on the ability to control water levels. In addition to water depth, vegetative structure and pattern can strongly influence species diversity. Consequently, wetland management frequently involves the manipulation of landform and/or successional stages of plant communities.

In the past, wetland-management activities were often directed toward the needs of featured species such as waterfowl. Wetland management has slowly changed from a featured-species approach to a community-oriented approach that strives to provide benefits to a maximum number of species. However, very little is known about the life history and habitat requirements of many nongame species. Moreover, the total number of species of smaller organisms (e.g., insects and bacteria) is not even known, much less the effect management activities have on these species. Consequently, intensive management for biodiversity is a difficult process. On the other hand, ecological principles that apply to populations and communities of game species also apply to most nongame species. Thus, many nongame species should benefit from wetland-management activities designed to help game species such as waterfowl and furbearing mammals.

The science of wetland management has moved from trial-and-error to development of technical skills and then to theory. Despite these advances, wetland management cannot be reduced to a series of rigid guidelines because most wetlands are dynamic, complex ecosystems. Consequently, managers must be creative and use their onsite expertise to develop flexible-management strategies. These strategies should be based on sound ecological principles and realistic management objectives. This requires a manager who (a) recognizes the seasonal needs of the flora and fauna, (b) knows the ecology of the local-wetland

ecosystem and adjacent uplands, and (c) applies appropriate principles to develop methods suitable for the local situation (Kadlec and Smith 1992). This task is becoming more difficult, however, because wetland managers are being asked to place more emphasis on biodiversity conservation, while simultaneously maintaining other wetland functions and values. To effectively meet this challenge, resource personnel must develop a better understanding of wetland ecology, as well as the life-history requirements of species that rely on wetlands.

Consideration must not only be given to charismatic species (e.g., game, threatened, and endangered species) but also other organisms associated with wetland ecosystems. Providing the required resources for such a diverse group of target species may be one of the biggest challenges facing wetland managers today (Laubhan and Fredrickson 1993). However, biodiversity conservation involves more than increasing the number of species on individual wetlands. The focus of management must change from individual wetlands and featured species to a larger scale approach that considers wetland complexes, associated uplands, habitat corridors, and ecological processes. This new challenge will require a comprehensive, integrated approach to wetland management. For example, a combination of management, restoration, and creation techniques may be required to restore and maintain the natural diversity of a local landscape, including wetland complexes.

In addition, carefully planned monitoring and assessment studies are needed to guide management strategies and test predictions about wetland ecology. This is an important concept because an adaptive-management strategy combined with experimental studies is needed to advance our wetland-management skills and make management a more predictable science. At the very least, each management operation should be treated as part of a long-term research program with preliminary observations, records of actions taken, and a follow-up measuring success (Weller 1987). Furthermore, wetland management should be viewed as an ongoing process in which life-history characteristics and habitat requirements of many species are unknown. Hence, managers must be prepared to incorporate information on a species' biology and habitat requirements into their management plans when it becomes available.

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Appendix A

Common and Scientific Names of Plants Named in Text

Alphabetical by Common Name

| Common name | Scientific name |
|--------------------------------------|----------------------------------|
| Alkali (saltmarsh) bulrush | <i>Scirpus maritimus</i> |
| Arrowheads | <i>Sagittaria</i> spp. |
| Asters | <i>Aster</i> spp. |
| Baldcypress (cypress) | <i>Taxodium distichum</i> |
| Barnyard grass | <i>Echinochloa</i> sp. |
| Beakrush | <i>Rhynchospora</i> sp. |
| Beggarticks | <i>Bidens</i> spp. |
| Black willow | <i>Salix nigra</i> |
| Blunt spikerush | <i>Eleocharis obtusa</i> |
| Broomsedge bluestem | <i>Andropogon virginicus</i> |
| Bulrush | <i>Scirpus</i> sp. |
| Buttonbush, Common | <i>Cephalanthus occidentalis</i> |
| Cattail | <i>Typha</i> sp. |
| Cedar, Atlantic White | <i>Chamaecyparis thyoides</i> |
| Cedar, Northern White | <i>Thuja occidentalis</i> |
| Cherrybark oak | <i>Quercus falcata</i> |
| Chufa flatsedge | <i>Cyperus esculentus</i> |
| Cocklebur | <i>Xanthium strumarium</i> |
| Coontails | <i>Ceratophyllum</i> spp. |
| Cordgrass | <i>Spartina</i> sp. |
| Cottonwoods | <i>Populus</i> spp. |
| Crabgrass | <i>Digitaria</i> sp. |
| Curltop ladysthumb | <i>Polygonum lapathifolium</i> |
| Duckweeds | <i>Lemna</i> spp. |
| Foxtail grasses | <i>Setaria</i> spp. |
| Hardstem bulrush | <i>Scirpus acutus</i> |

| Common name | Scientific name |
|------------------------------------|----------------------------------|
| Hickory | <i>Carya</i> sp. |
| Indigobush amorpha | <i>Amorpha fruticosa</i> |
| Japanese millet | <i>Echinochloa crusgalli</i> |
| Joe-pye-weed | <i>Eupatorium serotinum</i> |
| Larch (Tamarack) | <i>Larix laricina</i> |
| Lotus, American | <i>Nelumbo lutea</i> |
| Mangrove, Red | <i>Rhizophora mangle</i> |
| Marshpurslane | <i>Ludwigia</i> sp. |
| Marsh (swamp) smartweed | <i>Polygonum hydropiperoides</i> |
| Millet, wild (Walter's) | <i>Echinochloa walteri</i> |
| Moss | <i>Fontinalis</i> spp. |
| Needlerush | <i>Juncus roemerianus</i> |
| Nuttall oak | <i>Quercus nuttallii</i> |
| Oaks | <i>Quercus</i> spp. |
| Overcup oak | <i>Quercus lyrata</i> |
| Panic grass | <i>Panicum</i> sp. |
| Pennsylvania smartweed | <i>Polygonum pensylvanicum</i> |
| Phragmites (common reed) | <i>Phragmites australis</i> |
| Pin oak | <i>Quercus palustris</i> |
| Pines | <i>Pinus</i> spp. |
| Pondweeds | <i>Potamogeton</i> spp. |
| Purple loosestrife | <i>Lythrum salicaria</i> |
| Ragweed, Common | <i>Ambrosia artemisifolia</i> |
| Redroot | <i>Lachnanthes</i> sp. |
| Redroot flatsedge | <i>Cyperus erythrorhizos</i> |
| Rushes | <i>Juncus</i> spp. |
| Sago pondweed | <i>Potamogeton pectinatus</i> |
| Saltmarsh cordgrass | <i>Spartina alterniflora</i> |
| Sawgrass | <i>Cladium jamaicense</i> |
| Sedges | <i>Carex</i> spp. |
| Smartweeds | <i>Polygonum</i> spp. |
| Spatterdock | <i>Nuphar</i> sp. |
| Spikerush | <i>Eleocharis</i> sp. |
| Sprangletop | <i>Leptochloa panicoides</i> |
| Spruces | <i>Picea</i> spp. |
| Threesquare, common | <i>Scirpus americanus</i> |
| Tooth-cup | <i>Ammannia coccinea</i> |
| Trumpetcreeper | <i>Campsis radicans</i> |
| Tupelo, black | <i>Nysaa aquatica</i> |
| Water hyacinth | <i>Eichhornia crassipes</i> |
| Water oak | <i>Quercus nigra</i> |

| Common name | Scientific name |
|----------------------|-------------------------------|
| Water lilies | <i>Nymphaea</i> spp. |
| Water milfoils | <i>Myriophyllum</i> spp. |
| Watershield | <i>Brasenia schreberi</i> |
| Whitetop | <i>Scholochloa festucacea</i> |
| Widgeon grass | <i>Ruppia maritima</i> |
| Willow oak | <i>Quercus phellos</i> |
| Willows | <i>Salix</i> spp. |

Alphabetical by Scientific Name

| Scientific name | Common name |
|--|-------------------------|
| <i>Ambrosia artemisifolia</i> | Common ragweed |
| <i>Ammannia coccinea</i> | Tooth-cup |
| <i>Amorpha fruticosa</i> | Indigobush amorpha |
| <i>Andropogon virginicus</i> | Broomsedge bluestem |
| <i>Aster</i> spp. | Asters |
| <i>Bidens</i> spp. | Beggarticks |
| <i>Brasenia schreberi</i> | Watershield |
| <i>Campsis radicans</i> | Trumpet creeper |
| <i>Carex</i> spp. | Sedges |
| <i>Carya</i> sp. | Hickory |
| <i>Cephalanthus occidentalis</i> | Common buttonbush |
| <i>Ceratophyllum</i> spp. | Coontails |
| <i>Chamaecyparis thyoides</i> | Atlantic white cedar |
| <i>Cladium jamaicense</i> | Sawgrass |
| <i>Cyperus erythrorhizos</i> | Redroot flatsedge |
| <i>Cyperus esculentus</i> | Chufa flatsedge |
| <i>Digitaria</i> sp. | Crabgrass |
| <i>Eichhornia crassipes</i> | Water hyacinth |
| <i>Echinochloa</i> sp. | Barnyard grass |
| <i>Echinochloa crusgalli</i> | Japanese millet |
| <i>Echinochloa walteri</i> | Wild or Walter's millet |
| <i>Eleocharis</i> sp. | Spikerush |
| <i>Eleocharis obtusa</i> | Blunt spikerush |
| <i>Eupatorium serotinum</i> | Joe-pye weed |
| <i>Fontinalis</i> spp. | Moss |
| <i>Juncus</i> spp. | Rushes |
| <i>Lachnanthes</i> sp. | Redroot |
| <i>Larix laricina</i> | Larch (Tamarack) |

| Scientific name | Common name |
|--|----------------------------|
| <i>Leersia oryzoides</i> | Rice (giant) cutgrass |
| <i>Lemna</i> spp. | Duckweeds |
| <i>Leptochloa panicoides</i> | Sprangletop |
| <i>Ludwigia</i> sp. | Marshpurslane |
| <i>Lythrum salicaria</i> | Purple loosestrife |
| <i>Myriophyllum</i> spp. | Water milfoils |
| <i>Nelumbo lutea</i> | American lotus |
| <i>Nuphar</i> sp. | Spatterdock |
| <i>Nymphaea</i> spp. | Water lilies |
| <i>Nysaa aquatica</i> | Black tupelo |
| <i>Panicum</i> sp. | Panic grass |
| <i>Phragmites australis</i> | Phragmites (common reed) |
| <i>Picea</i> spp. | Spruces |
| <i>Pinus</i> spp. | Pines |
| <i>Polygonum</i> spp. | Smartweeds |
| <i>Polygonum hydropiperoides</i> | Marsh (swamp) smartweed |
| <i>Polygonum lapathifolium</i> | Curltop ladythumb |
| <i>Polygonum pensylvanicum</i> | Pennsylvania smartweed |
| <i>Populus</i> spp. | Cottonwoods |
| <i>Potamogeton</i> spp. | Pondweeds |
| <i>Potamogeton pectinatus</i> | Sago pondweed |
| <i>Quercus</i> spp. | Oaks |
| <i>Quercus falcata</i> | Cherrybark oak |
| <i>Quercus lyrata</i> | Overcup oak |
| <i>Quercus nigra</i> | Water oak |
| <i>Quercus nuttallii</i> | Nuttall oak |
| <i>Quercus palustris</i> | Pin oak |
| <i>Quercus phellos</i> | Willow oak |
| <i>Rhizophora mangle</i> | Red mangrove |
| <i>Rhynchospora</i> sp. | Beakrush |
| <i>Ruppia maritima</i> | Widgeon grass |
| <i>Sagittaria</i> spp. | Arrowheads |
| <i>Salix</i> spp. | Willows |
| <i>Salix nigra</i> | Black willow |
| <i>Scholochloa festucacea</i> | Whitetop |
| <i>Scirpus</i> sp. | Bulrush |
| <i>Scirpus acutus</i> | Hardstem bulrush |
| <i>Scirpus americanus</i> | Threesquare, common |
| <i>Scirpus maritimus</i> | Alkali (saltmarsh) bulrush |
| <i>Setaria</i> spp. | Foxtail grasses |
| <i>Spartina</i> sp. | Cordgrass |

| Scientific name | Common name |
|------------------------------------|-----------------------|
| <i>Spartina alterniflora</i> | Saltmarsh cordgrass |
| <i>Taxodium distichum</i> | Baldcypress (cypress) |
| <i>Thuja occidentalis</i> | Northern white cedar |
| <i>Typha</i> sp. | Cattail |
| <i>Xanthium strumarium</i> | Cocklebur |

Appendix B

Common and Scientific Names of Animals Named in Text

Alphabetical by Common Name

| Common name | Scientific name |
|--------------------------------|--------------------------------------|
| Alligator | <i>Alligator mississippiensis</i> |
| American bittern | <i>Botaurus lentiginosus</i> |
| Bald eagle | <i>Haliaeetus leucocephalus</i> |
| Beaver | <i>Castor canadensis</i> |
| Beetles | Coleoptera (order) |
| Black bear | <i>Ursus americanus</i> |
| Blackbird, Red-winged | <i>Agelaius phoeniceus</i> |
| Blackbird, Yellow-headed | <i>Xanthocephalus xanthocephalus</i> |
| Bobolink | <i>Dolichonyx oryzivorus</i> |
| Caddis flies | Trichoptera (order) |
| Canada goose | <i>Branta canadensis</i> |
| Carp | <i>Cyprinus carpio</i> |
| Cliff swallow | <i>Hirundo pyrrhonota</i> |
| Common loon | <i>Gavia immer</i> |
| Coot, American | <i>Fulica americana</i> |
| Crayfish | <i>Procambarus</i> spp. |
| Dabbling ducks | Anatini (tribe) |
| Deer | <i>Odocoileus</i> spp. |
| Diving ducks | Aythini (tribe) |
| Everglade kite | <i>Rostrhamus sociabilis</i> |
| Fish | Osteichthyes (class) |
| Flies and Midges | Diptera (order) |
| Florida panther | <i>Felis concolor coryi</i> |
| Frogs | Salientia (order) |
| Gallinule, Common | <i>Gallinula chloropus</i> |
| Gallinule, Purple | <i>Porphyryula martinica</i> |
| Grackle | <i>Quiscalus</i> sp. |

| Common name | Scientific name |
|--------------------------------|----------------------------------|
| Grebes | Podicipedidae (family) |
| Hérons and Egrets | Ardeidae (family) |
| Hooded merganser | <i>Lophodytes cucullatus</i> |
| Killdeer | <i>Charadrius vociferus</i> |
| King rail | <i>Rallus elegans</i> |
| Kingfisher, Belted | <i>Ceryle alcyon</i> |
| Mallard | <i>Anas platyrhynchos</i> |
| Marsh wren (long-billed) | <i>Cistothorus palustris</i> |
| Mayflies | Ephemeroptera (order) |
| Meadowlark | <i>Sturnella</i> sp. |
| Mink | <i>Mustela vison</i> |
| Muskrat | <i>Ondatra zibethicus</i> |
| Night-Heron | <i>Nycticorax</i> sp. |
| Nutria | <i>Myocaster coypus</i> |
| Oriole, Northern | <i>Icterus galbula</i> |
| Osprey | <i>Pandion halieatus</i> |
| Pelican, American white | <i>Pelicanus erythrorhynchos</i> |
| Plovers | Charadriidae (family) |
| Prothonotary warbler | <i>Protonotaria citrea</i> |
| Rails | Rallidae (family) |
| Rainbow trout | <i>Oncorhynchos mykiss</i> |
| Red-shouldered hawk | <i>Buteo lineatus</i> |
| Redhead | <i>Aythya americana</i> |
| Ruddy duck | <i>Oxyura jamaicensis</i> |
| Ruffed grouse | <i>Bonasa umbellus</i> |
| Sedge wren | <i>Cistothorus platensis</i> |
| Shorebirds | Charadriiformes (order) |
| Snails | Gastropoda (class) |
| Snakes | Serpentes (suborder) |
| Song sparrow | <i>Melospiza melodia</i> |
| Sora | <i>Porzana carolina</i> |
| Stoneflies | Plecoptera (order) |
| Swallows | Hirundinidae (family) |
| Swans | <i>Cygnus</i> spp. |
| Tanagers | <i>Piranga</i> spp. |
| Tern, Black | <i>Chlidonias niger</i> |
| Tern, Forester's | <i>Sterna forsteri</i> |
| Turtles | Testudines (order) |
| Upland sandpiper | <i>Bartramia longicauda</i> |
| Virginia rail | <i>Rallus limicola</i> |
| Waders | Ciconiiformes (order) |

| Common name | Scientific name |
|-------------------|--------------------|
| Wood duck | <i>Aix sponsa</i> |
| Woodpeckers | Piciformes (order) |

Alphabetical by Scientific Name

| Scientific name | Common name |
|---|--------------------------|
| <i>Agelaius phoeniceus</i> | Red-winged blackbird |
| <i>Aix sponsa</i> | Wood duck |
| <i>Alligator mississippiensis</i> | Alligator |
| <i>Anas platyrhynchos</i> | Mallard |
| Anatini (tribe) | Dabbling ducks |
| Ardeidae (family) | Hérons and Egrets |
| Aythini (tribe) | Diving ducks |
| <i>Aythya americana</i> | Redhead |
| <i>Bartramia longicauda</i> | Upland sandpiper |
| <i>Bonasa umbellus</i> | Ruffed grouse |
| <i>Botaurus lentiginosus</i> | American bittern |
| <i>Branta canadensis</i> | Canada goose |
| <i>Buteo lineatus</i> | Red-shouldered hawk |
| <i>Castor canadensis</i> | Beaver |
| <i>Ceryle alcyon</i> | Belted kingfisher |
| Charadriiformes (order) | Shorebirds |
| Charadriidae (family) | Plovers |
| <i>Charadrius vociferus</i> | Killdeer |
| <i>Chlidonias niger</i> | Black tern |
| Ciconiiformes (order) | Waders |
| <i>Cistothorus palustris</i> | Marsh wren (long-billed) |
| <i>Cistothorus platensis</i> | Sedge wren |
| Coleoptera (order) | Beetles |
| <i>Cygnus</i> spp. | Swans |
| <i>Cyprinus carpio</i> | Carp |
| Diptera (order) | Flies and Midges |
| <i>Dolichonyx oryzivorus</i> | Bobolink |
| Ephemeroptera (order) | Mayflies |
| <i>Felis concolor coryi</i> | Florida panther |
| <i>Fulica americana</i> | American coot |
| <i>Gallinula chloropus</i> | Common gallinule |
| Gastropoda (class) | Snails |
| <i>Gavia immer</i> | Common loon |
| <i>Haliaeetus leucocephalus</i> | Bald eagle |

| Scientific name | Common name |
|--|-------------------------|
| Hirundinidae (family) | Swallows |
| <i>Hirundo pyrrhonota</i> | Cliff swallow |
| <i>Icterus galbula</i> | Northern oriole |
| <i>Lophodytes cucullatus</i> | Hooded merganser |
| <i>Maleagris gallopavo</i> | Turkey |
| <i>Melospiza melodia</i> | Song sparrow |
| <i>Mustela vison</i> | Mink |
| <i>Myocaster coypus</i> | Nutria |
| <i>Nycticorax</i> sp. | Night-Heron |
| <i>Odocoileus</i> spp. | Deer |
| <i>Oncorhynchos mykiss</i> | Rainbow trout |
| <i>Ondatra zibethicus</i> | Muskrat |
| Osteichthyes (class) | Fish |
| <i>Oxyura jamaicensis</i> | Ruddy duck |
| <i>Pandion halieatus</i> | Osprey |
| <i>Pelicanus erythrorhynchos</i> | American white pelican |
| Piciformes (order) | Woodpeckers |
| <i>Piranga</i> spp. | Tanagers |
| Plecoptera (order) | Stoneflies |
| Podicipedidae (family) | Grebes |
| <i>Porphyryla martinica</i> | Purple gallinule |
| <i>Porzana carolina</i> | Sora |
| <i>Procambarus</i> spp. | Crayfish |
| <i>Protonotaria citrea</i> | Prothonotary warbler |
| <i>Quiscalus</i> sp. | Grackle |
| Rallidae (family) | Rails |
| <i>Rallus elegans</i> | King rail |
| <i>Rallus limicola</i> | Virginia rail |
| <i>Rostrhamus sociabilis</i> | Everglade kite |
| Salientia (order) | Frogs |
| <i>Sterna forsteri</i> | Forester's tern |
| <i>Sturnella</i> sp. | Meadowlark |
| Testudines (order) | Turtles |
| Trichoptera (order) | Caddis flies |
| <i>Ursus americanus</i> | Black bear |
| <i>Xanthocephalus xanthocephalus</i> | Yellow-headed blackbird |

Appendix C

Selected Readings

The following is a listing of selected readings on 12 topics related to marsh management and biodiversity conservation.¹

Freshwater Marshes: Ecological Processes and Biophysical Characteristics

Cross 1988 (sec. 13.3)
Good, Whigham, and Simpson 1978
Hook et al. 1988a,b
Mitsch and Gosselink 1993
van der Valk 1989
Weller 1987

Principles of Landscape Ecology and Ecosystem Management

Forman and Gordon 1981
Franklin 1993
Hudson 1991
Noss 1983
Temple 1986
Urban, O'Neill, and Shugart 1987

Impacts to Wetlands and Wetland Wildlife

Cairns 1990
Dahl and Johnson 1991
Fredrickson and Reid 1990
Harris 1988
Niering 1988
OTA 1984
Tiner 1984
Weller 1988

¹ References cited in this appendix are located at the end of the main text.

Planning, Assessment, and Monitoring

Cooperrider, Boyd, and Stuart 1986
Fredrickson and Laubhan 1994
Macnab 1983
Payne 1992
Ratti and Garton 1994
Weller 1986

Artificial Nesting and Loafing Structures

Ball 1990
Linde 1969
Lokemoen and Messmer 1994
Mackey, Mathews, and Ball 1988
Payne 1992
Yoakum et al. 1980

Wetland Restoration and Creation

Cairns 1988
Hammer 1992
Kentula et al. 1992
Kusler and Kentula 1990
NRC 1992a
Schneller-McDonald, Ischinger, and Auble 1990

Biodiversity and Conservation Biology: Theory, Principles, and Management

Cairns and Lackey 1992
Knopf and Smith 1992
Meffe and Carroll 1994
Murphy 1989
OTA 1987
Soulé 1986
Wilson 1988

Wetland Management: General Guidelines and Techniques

Fredrickson and Laubhan 1994
Kadlec and Smith 1992
Linde 1969
Payne 1992
Smith, Pederson, and Kaminski 1989
Weller 1982, 1987

Wetland Management for Shorebirds and Other Species

Brown and Dinsmore 1986
Clark 1993
Finney and Castro 1993
Fredrickson and Reid 1986
Fredrickson and Taylor 1982
Helmert 1992
Knighton 1985
Wentz and Reid 1992

Integrated Wetland Management

Fredrickson and Reid 1986
Laubhan and Fredrickson 1993

Control of Undesirable Species

Cross and Fleming 1989
Fredrickson and Reid 1988b
Linde 1969
Payne 1992
Thompson 1989

Revegetation Strategies and Techniques

Allen et al. 1989
Kadlec and Wentz 1974
Marburger 1992
Reinartz and Warne 1993
Thunhorst 1993

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| 4.TITLE AND SUBTITLE Interior Wetlands of the United States: A Review of Wetland Status, General Ecology, Biodiversity, and Management | | | 5.FUNDING NUMBERS | |
| 6.AUTHOR(S) John H. Giudice and John T. Ratti | | | | |
| 7.PERFORMING ORGANIZATION NAME(S) AND ADDRESS(ES) Department of Fish and Wildlife Resources University of Idaho Moscow, ID 83844-1136 | | | 8.PERFORMING ORGANIZATION REPORT NUMBER | |
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| 12a.DISTRIBUTION/AVAILABILITY STATEMENT Approved for public release; distribution is unlimited. | | | 12b.DISTRIBUTION CODE | |
| 13.ABSTRACT (Maximum 200 words) <p>Ecosystem management and the conservation of biodiversity has become an important public-policy issue in the United States. Although national attention has focused on terrestrial ecosystems, wetlands are an important component in efforts to conserve biodiversity. Freshwater wetlands support a diversity of plant and animal species, including a third of the nation's threatened and endangered species. Wetland managers are being asked to place more emphasis on biodiversity and natural community characteristics, while simultaneously maintaining other wetland functions and values. To effectively meet this challenge, wetland managers will need to understand the principles and concepts of conservation biology and ecosystem management.</p> <p>This report was compiled to provide Corps field and District-level personnel with a primer on the ecology, biodiversity, and management of freshwater wetlands in the United States. An overview is provided of the principles, concepts, strategies, and techniques necessary to preserve, restore, create, and manage natural community and biodiversity characteristics of nontidal, freshwater wetlands. These interior wetlands include marshes, potholes, swamps, bogs, fens, and riparian systems. General information is also provided on wetland definitions, classification and inventory, status and distribution, general ecology, functions and values, and programs affecting wetland conservation.</p> | | | | |
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